INTEGRATING HISTORICAL IMAGERY AND SEDIMENT RADIOISOTOPES TO SHED LIGHT ON LONG-TERM RANGELAND DYNAMICS AND ECOSYSTEM SERVICES AT THE WATERSHED SCALE

A Dissertation

by

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ABSTRACT

Rangelands are widespread and include grasslands, shrublands, and woodlands, covering more than two thirds of the earth's land surface. This cover type represents half of the land area in the United States. These systems are extremely dynamic and respond to natural processes as well as varying degrees of human impact. In the southern Great Plains, land cover and land use have changed dramatically over the last century. However, there is tremendous uncertainty as to the timing and magnitude of such transitions and much more so regarding the effects on hydrology and sediment dynamics in these areas. Using a watershed approach in the Lampasas Cut Plain of Texas, we applied object-oriented classification methods and hand-digitizing of historical aerial photos to track the extent of woody plant cover, cropland area, and small ponds through time. We compared these results with population trends to determine the relationship between social and environmental variables. Finally, we conducted sediment analyses of cores from constructed reservoirs in each watershed to establish a chronological sequence of rangeland processes.

Woody plant cover displayed very complex responses between areas yet was very similar among watersheds in the same setting. Prolonged decreases occurred over the first half of the study period before rebounding in Lampasas County. By contrast, shrub cover decreased and remained low and stable in Mills County over the last several decades. Woody cover in an urbanized watershed consistently increased. Cropland area showed marked decreases over all areas, declining by 77% between the 1930s and 2012.

The number of small ponds increased by over 250% over the same period. Trends in woody plant cover were closely related to population in each context. The opposing trends of cropland and pond density were strongly correlated. Precipitation, streamflow, and baseflow were largely unchanged over the last 90 years, suggesting a minimal impact of land use and land cover on local hydrology. Sediment delivery did increase immediately after drought periods, when intense rainfall caused soil loss as a result of drought-induced vegetation loss. These findings are critical to understanding the implications for future sustainability of rangeland landscapes and the ecosystem services they provide.

DEDICATION

To those who strive to never stop learning from the past, who relish anecdotes yet press on for more evidence, and who strive to see the big picture while constantly poring over the details.

ACKNOWLEDGEMENTS

This dissertation is entirely a team effort and nothing less. Without the intense image processing effort put forth by Jay Angerer and Ed Rhodes, the already lengthy image classification process would have stretched many times longer. Without the training provided by Sorin Popescu, I wouldn't even have known where to start with remote sensing. Without the mechanically minded tinkering and deep investment of time by Jason McAlister, the sediment coring would have been impossible. Without the dedicated isotope analyses made possible by Mead Allison and his teams at the University of Texas and Tulane University, I would still have buckets of pond mud I wouldn't know what to do with. Without the experience and input of Franco Marcantonio, charts on a page would provide zero information. Without the hydrological assistance of Steve Potter, I would be stuck guessing on streamflow trends. In addition to the immeasurable academic support, I have received I could not have survived without the encouragement and patience of my friends, roommates, and family who provided affirmation at each of the moments it was needed most. Finally, I am indebted to Brad Wilcox for his generous investment in me throughout my time in the Department of Ecosystem Science and Management. He stood behind all of my endeavors, even those that didn't directly benefit our research and those that made no sense at all, and provided countless opportunities to lead and make a name for myself in the science and management of natural resources. I have been blessed with a trusting and supportive adviser, supervisor, and friend.

NOMENCLATURE

APFO	Aerial Photography Field Office
cm	Centimeters
CRP	Conservation Reserve Program
ETM+	Enhanced Thematic Mapper Plus
FCS	Flood Control Structure
GIS	Geographic Information System
HPGe	High-Purity Germanium
km	Kilometers
m	Meters
mm	Millimeters
NAIP	National Agriculture Imagery Program
Q	Streamflow
Q_b	Baseflow
Q_p	Peak Flow
PMDI	Palmer Modified Drought Severity Index
SVM	Support Vector Machine
TNRIS	Texas Natural Resource Information System
TxDOT	Texas Department of Transportation
USDA	United States Department of Agriculture
USGS	United States Geological Survey

TABLE OF CONTENTS

Page

LIST OF FIGURES

Page

Page

Page

LIST OF TABLES

CHAPTER I

INTRODUCTION

Dominated by herbaceous plants but including a wide array of plant species and growth forms and falling across a spectrum of precipitation and temperature gradients, rangelands are a major component of the planet's land surface. These areas are key regions for the production of livestock, serve as important wildlife habitat, and offer a variety of valuable ecosystem services that are depended upon by human populations, even those hundreds of miles away. But in many cases, these areas face numerous challenges and changes associated with growing populations, changing climates, invasive species, shifting management priorities, and other stressors.

One of the most apparent changes in rangelands is the phenomenon of woody plant encroachment. Though woody plants (shrubs and trees) are important components of many rangeland ecosystems, observations indicate that their relative abundance and total coverage has increased in recent decades, changing open savannas and grasslands to shrublands and woodlands that pose major problems for historic uses of these lands. Higher concentrations of brushy species reduce the availability of forage and force animals to graze in increasingly confined areas. At the same time, livestock have been suggested as one of the key causes of the expansion of woody plant coverage through the distribution of seeds, removal of competing herbaceous species, and disruption of soil, creating sites that favor the establishment of woody species.

However, the timing and extent of this transition is poorly understood. A growing number of studies have examined changes in shrub cover at relatively small scales, assessing woody plant patches and performing field-scale analyses. Despite this growing body of knowledge, significant knowledge gaps remain at larger spatial and temporal scales. While field-scale studies can be helpful, the landscape context of plant community changes – and the corresponding effects on ecosystem processes – is of much greater concern. For this reason, using large data sets spanning broad areas and long time series can be a powerful tool in understanding when and where woody plant encroachment truly has occurred.

In addition to documenting actual transitions to woody plant-dominated landscapes, it is also possible – and insightful – to identify other landscape changes that have occurred in parallel with increasing shrub cover. The extent of cropland areas has undergone very distinct trends across the United States since the earliest periods of settlement. As new land areas were developed, crop production consumed an increasing proportion of local lands, largely related to human population. Eventually, after decades or centuries in production, many of these croplands were abandoned in favor of other land uses due to declines in productivity, shifts in land value and landowner priorities, and availability of virgin croplands in other regions. It has been suggested that many areas experiencing an increase in shrub cover are old croplands that favor woody plant establishment due to their high levels of disturbance. Thus, identifying the extent and distribution of these areas through time is a critical piece of the puzzle to understanding woody plant encroachment.

Similarly, small constructed ponds have become a major feature of many rangelands, though they are not natural in many of these landscapes. It is suspected that these both result from and cause land use changes in rural watersheds, facilitating and indicating a shift away from cropland focus to that of rangeland management.

While understanding the visible changes of plant community transformation, cropland abandonment, and pond construction is important, it is the effects of these shifts that are supremely important. Since rangelands often exist in subhumid climates, water often is at a premium, and improving our understanding of the hydrological characteristics of varying rangelands is vital to water resource planning now and into the future. A number of approaches support investigation into watershed dynamics, from long-term weather data and historical stream gages to sediment radioisotopes that can be used to estimate changing sedimentation rates over time as indicators of watershed soil loss and sediment yield. Pulling these sources together enables a more complete picture of watershed function under varying land use and land cover patterns through time. When each of these bodies of data is brought together, it may be possible to make land management recommendations to maximize natural resource availability, both locally and beyond.

CHAPTER II

LONG-TERM DYNAMICS OF WOODY PLANT COVER AND PRODUCTION AGRICULTURE IN THE LAMPASAS CUT PLAIN, TEXAS

Introduction

Dominated by grasslands prior to European settlement, the southern Great Plains of Texas and Oklahoma today are a complex landscape that includes both dryland and irrigated agriculture, rangelands, woodlands, and urban centers. This dynamic region faces many complex changes of the future, including land use change, crop production demand and policies, local and global economic conditions, population growth, water scarcity, climate change, and other challenges (Drummond *et al.*, 2012). Among these are changes in vegetation composition and cover. Instances of plant encroachment into more open landscapes dominated by herbaceous species have been documented over the last several decades around the world (Archer, 1994). Many of these observations are recorded from subhumid and semiarid regions, particularly in middle latitudes, with important effects on land use, biogeochemical cycling, and biodiversity. In many cases, these shrub invasions have been demonstrated as linked to ecosystem degradation. Previous concerns over this phenomenon have centered on effects of hydrology and production of livestock and wildlife, but the focus of an increasing number of investigations is on the implications of woody plant encroachment of carbon cycling, biodiversity, and erosion (Archer *et al.*, 2011). Yet management strategies largely still focus on those issues of immediate impact on surrounding populations. Increasing woody plant cover has been suggested as reducing water yield in semi-arid landscapes, with effects for local and downstream supplies (Bednarz *et al.*, 2000, Griffin & McCarl, 1989, Knight & Thurow, 1991, Texas State Soil and Water Conservation Board, 2005, Welch, 1991). However, evidence of this at large scales is quite rare. Invasion by *Juniperus* spp. of deciduous woodlands has also led to widespread conversion to evergreen woodlands with implications for ecological succession and wildlife (Coppedge *et al.*, 2001).

A number of mechanisms for increase in woody plant abundance have been proposed, chiefly including changes due to livestock grazing, altered fire regimes, and climate (Grover & Musick, 1990, Van Auken, 2000). In much of the southern Great Plains, the removal of fire from the ecosystem and subsequent conversion to production agriculture are suggested as primary drivers acting to produce a positive feedback mechanism favoring fragmentation and woody plant encroachment (Coppedge *et al.*, 2001). Lands throughout the region have been influenced by cropland agriculture and especially livestock production since the 1850s, leading to some concerns for long-term sustainability and ecosystem function even by the close of the 19th century (Bentley, 1898, Jordan-Bychkov *et al.*, 1984). However, multiple key questions remain, especially when considering larger spatial and temporal scales.

Analysis of time series aerial photography provides a powerful tool in determining historical changes in land cover over broad areas and long periods (Platt & Schoennagel, 2009). Sequential historical imagery analysis represents a series of snapshots of the landscape, each documenting conditions at a specific point in time.

Even among the oldest available imagery, minimal woody plant detection limits have been shown to be comparable to those for modern imagery (Browning *et al.*, 2009). This approach is particularly helpful in extracting ecological information from prior periods even where plot studies were not conducted, and early aerial imagery often is employed in setting local baselines of ecosystem structure (Laliberte *et al.*, 2004). While landscapes marked by strong episodic recruitment and mortality events relating to climatic variations or disturbance may require much more frequent examination, this strategy is very helpful in quantifying large-scale changes (Browning *et al.*, 2008).

Examinations of woody plant encroachment in Texas and other portions of the southwest United States have indicated widespread shrub increase throughout much of the 20th century (Ansley *et al.*, 1995, Asner, 2003, Laliberte *et al.*, 2004, Smeins & Merrill, 1988) as well as significant changes in cultivated cropland area over the same period (Hart, 1968). However, few studies have corporate both trends to synthesize a holistic view of landscape change. The present work intends to tell a complete story of land cover and land use change in central Texas since early parts of the last century. Our objectives in this study are to identify the timing and extent of 1) woody plant encroachment and 2) cropland abandonment trends at the watershed scale in eight watersheds in the Lampasas Cut Plain of Texas. We anticipate that total cover of trees and shrubs has increased dramatically over the last 80 years, likely at a higher rate in most recent decades. In addition, we suspect the area has experienced a consistent decrease in cropland coverage in the area, likely punctuated by accelerated rates of decline following the severe drought of the 1950s.

Materials and Methods

Location Description

For this study, we selected eight watersheds primarily within Lampasas and Mills Counties in central Texas, with a portion of two watersheds falling in Burnet County (Figure 2.1). The region is part of the Lampasas Cut Plain, a transitional zone between the prairies and Cross Timbers of north central Texas and the Edwards Plateau (Allison, 1991, Clower, 1980). This area, characterized by broad valleys separated by low buttes and mesas, is dominated by a mosaic of herbaceous and woody plant species. Annual precipitation ranges from 710 mm in parts of Mills County to near 780 mm in southern portions of the study area in northern Burnet County. The eight study watersheds enclose intermittent headwater streams that drain to reservoirs constructed as a result of the Watershed Protection and Flood Prevention Act of 1954. The watersheds are located in three distinct geographical clusters: 1) five adjoining watersheds in rural Lampasas County and northern Burnet County, 2) one small peri-urban watershed in the City of Lampasas, and 3) two adjoining watersheds in rural Mills County. With a total area of approximately 230 km^2, the watersheds range from 1.24 km^2 to 50.92 km^2. A record of the year of Flood Control Structure (FCS) construction and associated characteristics for the watershed of each site is provided in Table 2.1.

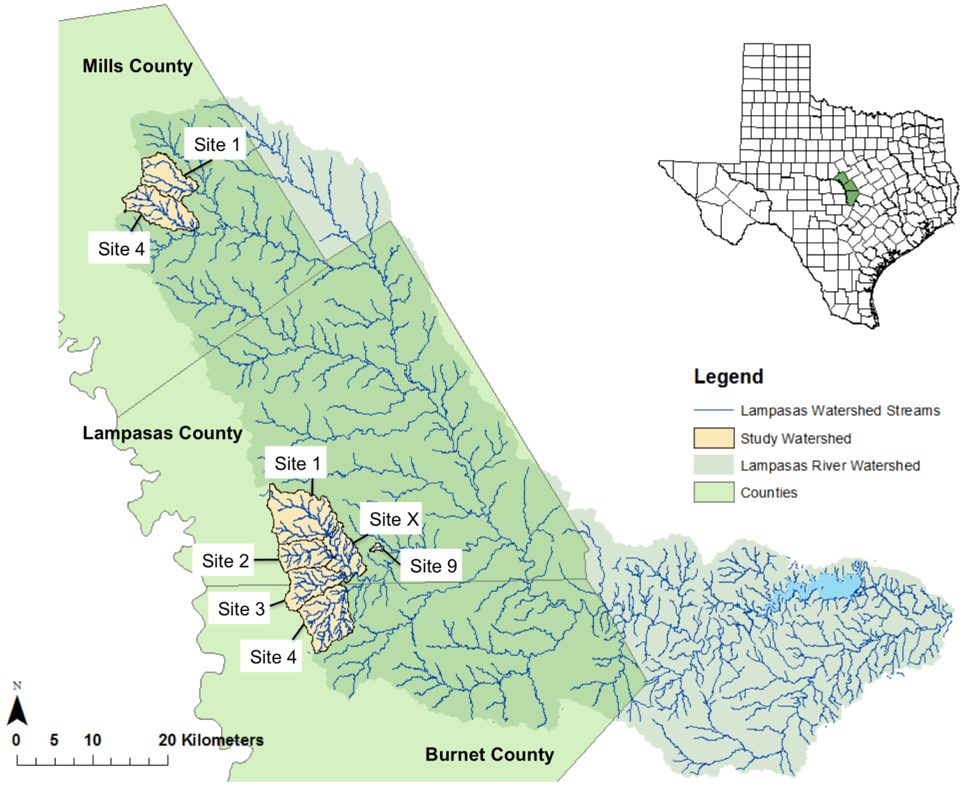


Figure 2.1. Map of study watershed location in central Texas.

Table 2.1. List of study watersheds and basic characteristics.

FCS Designation	Year Constructed	Watershed Area (km^2)	Maximum Elevation (m)	Minimum Elevation (m)
Lampasas Site 1	1959	50.92	500	367
Lampasas Site 2	1959	23.23	458	362
Lampasas Site 3	1958	27.49	459	355
Lampasas Site 4	1960	41.20	467	345
Lampasas Site 9	1960	1.24	363	322
Lampasas Site X	1961	23.11	420	338
Mills Site 1	1974	34.61	536	422
Mills Site 4	1974	27.98	534	432

Data Sources

We obtained aerial and satellite imagery from the earliest period available followed by approximately 8- to 15-year increments to the present. Images were provided by a number of sources, including the USDA-Aerial Photography Field Office (APFO), Texas Natural Resource Information System (TNRIS), National Agriculture Imagery Program (NAIP), and the private firm, Tobin International, Ltd. A description of imagery characteristics is found in Table 2.2. Due to varying availability for different portions of the study area, this resulted in imagery from 1937/1940/1941, 1958, 1974/1975, 1980/1982, 1995, 2004, and 2012. Late fall and winter images were preferred to take advantage of the contrast between predominantly evergreen woody plants and herbaceous species. Spatial resolution ranged from 1:1667 to 1:40000 and represented a combination of black and white, color infrared, and natural color imagery. Where necessary, hard copies were digitally scanned, and all images were radiometrically corrected and georectified to 2004 NAIP imagery (<1 m error). Following the procedure described by Laliberte *et al*. (2004). we then applied a 3 x 3 kernel low-pass filter to reduce spatial frequency. Finally, we resampled all images to a common 1 m resolution using nearest neighbor methodology to standardize analysis and maintain consistency in appearance of landscape features across resolutions from different images.

Table 2.2. Characteristics of remote sensing imagery for each image year.

Area	Image Year	Acquisition Date	Type	Spatial Scale	Spatial Resolution (m)
Lampasas Sites 1-4, X	1940	December 1939-August 1941	BW	1:1667	0.45
	1958	December 1957-January 1958	BW	1:20000	0.24-0.51
	1974	February 1974	BW	1:40000-1:85000	1.01-1.02
	1982	December 1982	BW	1:4700	0.90
	1995	January 1995-February 1995	CIR	1:40000	1.01
	2004	September 2004-December 2004	CIR	1:40000	1.00
	2012	October 2012	NC	1:12000	1.00
Lampasas Site 9	1941	August 1941	BW	1:1667	0.45
	1958	December 1957	BW	1:20000	0.24
	1974	February 1974	BW	1:85000	1.02
	1982	December 1982	BW	1:4700	0.90
	1995	February 1995	CIR	1:40000	1.01
	2004	December 2004	CIR	1:40000	1.00
	2012	October 2012	NC	1:12000	1.00
Mills Sites 1,4	1937	December 1937-Novermber 1938	BW	1:1667	0.40
	1958	December 1957-April 1959	BW	1:20000	0.51
	1975	November 1975	BW	1:60000	1.02
	1980	November 1980	BW	1:2640	0.63
	1995	January 1995	CIR	1:40000	1.01
	2004	December 2004	CIR	1:40000	1.00
	2012	October 2012	NC	1:12000	1.00

Woody Plant Coverage

Using the imagery above, we performed object-oriented image classification using the Example-Based Feature Extraction module with the SVM algorithm in the ENVI 5.0 software package (Exelis, 2013). This object-based approach is advantageous in that it segments imagery into pixel groups based on similar spectral, textural, and spatial characteristics, each with ecological meaning that can be interpreted with expert knowledge (Laliberte *et al.*, 2004, Platt & Schoennagel, 2009). We analyzed imagery for each of the eight watersheds from each year in the time series, with image segments designated as one of three classes: 1) herbaceous cover, 2) bare ground/road cover, and 3) woody plant cover. All objects were assigned to one of these categories. Bare ground and roadways were combined to reflect their similar characteristics, as the majority local rural roadways are unpaved and function much the same as bare ground. Woody plants included each evergreen coniferous, evergreen broadleaf, and deciduous broadleaf trees and shrubs, often identifying individual plants. While imagery from more recent years supports the separation of these groups, older imagery does not, due to spectral and seasonal coverage limitations (Browning *et al.*, 2009). Buildings and water classes were not included in the overall analysis, due to the difficulties in separating the varying appearances of structures and small ponds from each of the other classes in older imagery as well as being a very small proportion of overall watershed area. We hand-digitized FCS reservoir boundaries for each image to identify fluctuating water levels between years. Following image classification, we calculated the watershed area of each cover type for the entire time series to assess the magnitude and rate of change over time.

Cropland Trends

To understand land use dynamics and inform our interpretation of the land cover classification, we identified croplands in active use over time for each watershed using the imagery sources described previously. Agricultural fields were delineated based on the presence of either: 1) evidence of harvest activities or 2) evidence of site preparation activities. This allows for identification of fields in which crops are grown that do not correspond to the seasonal timing of imagery. In addition, fields with uniform texture and spectral signature distinct from adjoining rangelands were classified as croplands if they exhibited 1) zero evidence of encroachment by shrub saplings and 2) no sign of long-term grazing by livestock. As a result, hayfields were considered active croplands, while grazed rangelands were not. Appearance between seasons via Google Earth historical imagery was used to confirm cropland classification. Fields were hand-digitized and delineated using ArcGIS 10.0 (ESRI, 2010) for each imagery year for each watershed.

Results

Woody Plant Coverage

Herbaceous vegetation has been the dominant cover class over the past eight decades, with woody plants a key secondary cover type in the watersheds examined.

Some very distinct dynamics are apparent over that time. With respect to woody plant cover, we observe three general trends that correspond to groups of watersheds in similar landscape settings.

Watersheds of Lampasas Sites 1-4 as well as at Lampasas Site X, all in rural Lampasas County, displayed relatively high levels of woody plant cover in 1940-41, ranging from 16.7% to 27.1% (Table 2.3). This cover decreased gradually until the 1970s, reaching a minimum of 11.8% to 16.1%. After this period, woody vegetation began to increase and, by 2012, return to levels somewhat above (24.8% to 41.4%) those observed several decades earlier at the beginning of the study period. As a result, a V-shaped trend characterizes the long-term woody vegetation cover in these areas (Figure 2.2). Even within broad periods of woody plant increase, there are occasional intervals of much slower expansion during portions of the last 30 years, including between 1982 and 1995 and then again from 2004 to 2012. Conversely, the period of 1974-1982 typically marked the greatest increase in woody plant cover in these watersheds.

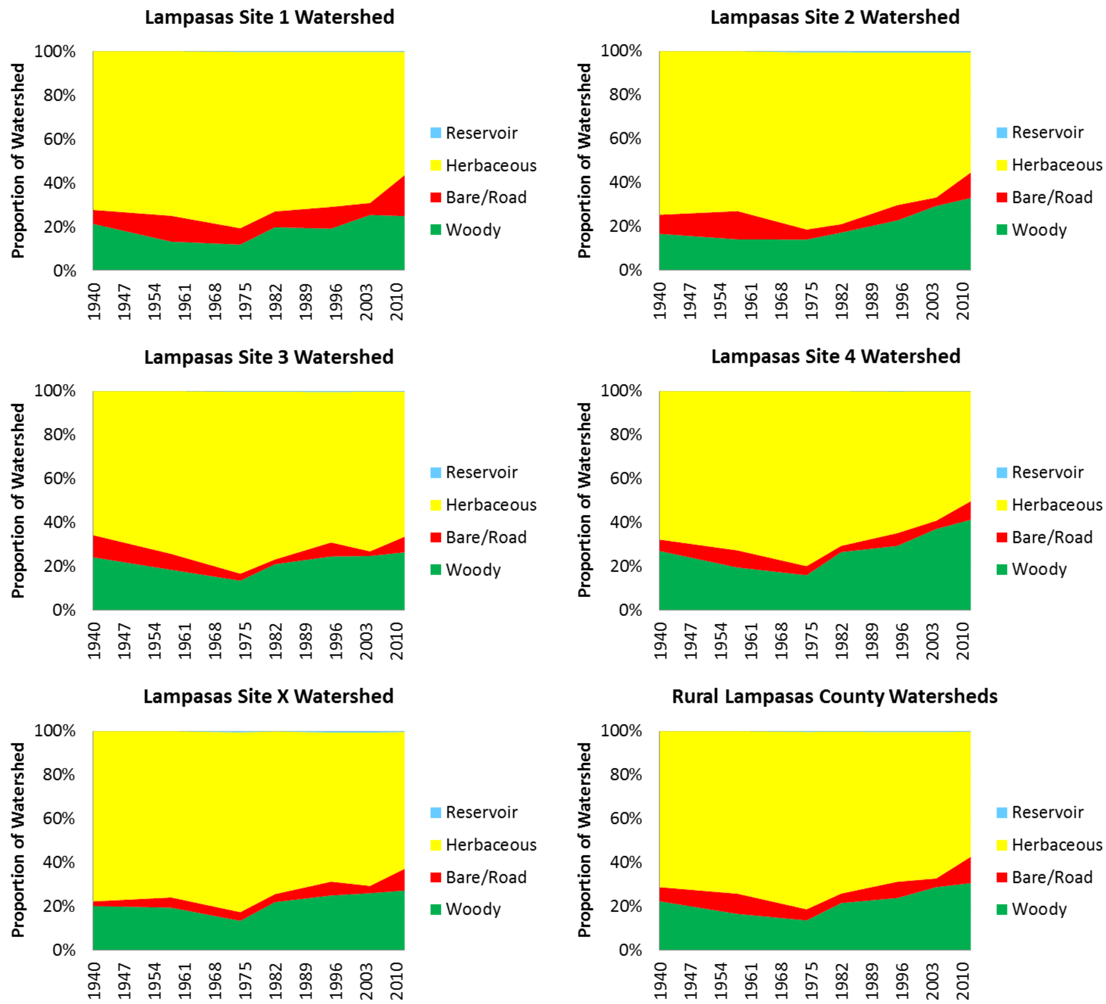


Figure 2.2. Changing land cover over time in rural Lampasas County watersheds. A summary chart combining all rural Lampasas watersheds is included. Each displays a flattened V-shaped trend in woody plant cover over the last 70 years.

Table 2.3. Woody plant cover by area and proportion of watershed area for each watershed over time.

		1937-41	1958	1974-75	1980-82	1995	2004	2012
Lampasas Site 1*	Area (km^2)	10.79	6.70	6.01	10.02	9.71	12.89	12.62
	%	21.2	13.2	11.8	19.7	19.1	25.3	24.8
Lampasas Site 2	Area (km^2)	3.88	3.29	3.27	4.01	5.30	6.83	7.68
	%	16.7	14.2	14.1	17.3	22.8	29.4	33.0
Lampasas Site 3*	Area (km^2)	6.60	5.10	3.73	5.75	6.77	6.83	7.29
	%	24.2	18.6	13.6	21.1	24.6	24.9	26.5
Lampasas Site 4	Area (km^2)	11.17	8.07	6.62	10.96	12.13	15.31	17.05
	%	27.1	19.6	16.1	26.6	29.4	37.2	41.4
Lampasas Site 9	Area (km^2)	0.17	0.20	0.26	0.40	0.40	0.43	0.40
	%	13.8	16.1	21.1	32.7	32.6	34.4	32.6
Lampasas Site X	Area (km^2)	4.68	4.51	3.21	5.09	5.80	6.03	6.31
	%	20.3	19.5	13.5	22.0	25.1	26.1	27.3
Lampasas Sites	Area (km^2)	37.38	27.88	23.03	36.31	40.12	48.33	51.36
	%	22.4	16.7	13.8	21.7	24.0	28.9	30.7
Mills Site 1	Area (km^2)	13.71	7.10	5.02	5.07	4.73	5.25	4.82
	%	39.6	20.5	14.5	14.7	13.7	15.2	13.9
Mills Site 4	Area (km^2)	14.16	6.60	5.43	5.90	5.63	5.31	4.76
	%	51.0	23.6	19.4	21.1	20.1	19.0	17.0
Mills Sites	Area (km^2)	27.87	16.70	10.45	10.97	10.36	10.56	9.59
	%	44.5	21.9	16.7	17.5	16.5	16.9	15.3
All Sites	Area (km^2)	65.25	41.58	33.48	47.28	50.48	58.89	60.94
	%	28.4	18.1	14.6	20.6	22.0	25.6	26.5

*Due to image availability, very minor portions of coverage from 1940 and 1982 for Lampasas Sites 1 and 3 were omitted. In each case, the omitted area is less than 1% of the total watershed area. Image classification was performed for remaining portion of watershed.

Lampasas Site 9, the small watershed partially within the City of Lampasas, exhibits an upward trend in woody plant cover, with slow expansion preceding an accelerated increase followed by a somewhat steady state in total cover (Figure 2.3). Cover increased from 13.8% in 1941 to 32.7% in 1982, with nearly identical total cover in 2012 at 32.6%. As in the rural Lampasas County watersheds the period of most rapid woody plant cover expansion at Lampasas Site 9 occurred between 1974 and 1982. A common trend in all watersheds is the sizable increase in the bare ground/road class in drought periods, and this is especially apparent in 2012 (Figure 2.4). Significant development occurred in this watershed over the study period, with a number of new roads and subdivisions constructed, particularly in the southern and central portions of the watershed. The gridlike feature running from northwest to southeast in the southeast of the area is the municipal cemetery, which has been a prominent member of the watershed since before 1940. Though the net effect is an increase in woody plant cover, there was a great deal of change between years in different portions of the watershed, as some plots experienced significant brush management activities.

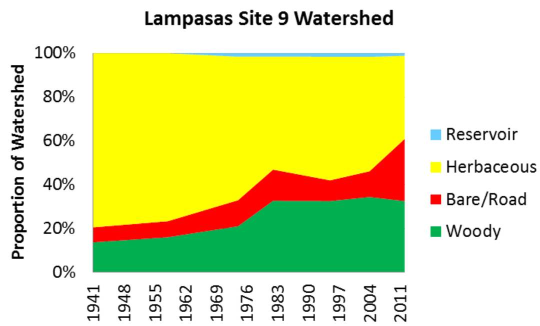


Figure 2.3. Changing land cover over time in peri-urban Lampasas Site 9 watershed, partially within city limits. This watershed shows a gradual increase in woody plant cover over the last 70, with little change over the last three decades.

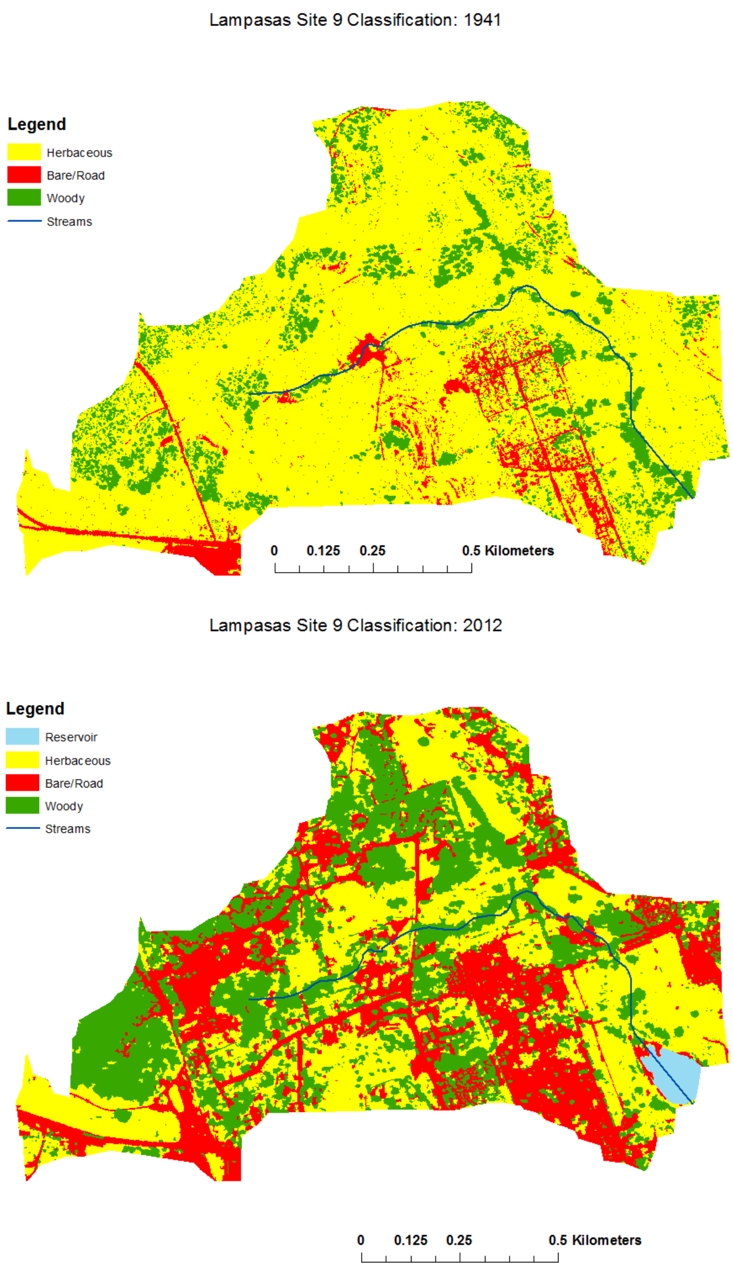


Figure 2.4. Comparison of land cover in Lampasas Site 9 watershed between 1941 (top) and 2012 (bottom). Woody plant cover has increased and become more dense, particularly to the north and west. Increases in red are related to greater amounts of urban development and bare soil resulting from intense drought.

Mills Sites 1 and 4 each displayed a higher degree of woody plant cover at the start of the study period (39.6% to 51.0%) than did watersheds in Lampasas County. After this time, a downward trend in total plant cover followed, with initial rapid reduction followed by slow reduction or minimal change beginning about 1975 and extending for the next three decades (Figure 2.5). Between 2004 and 2012, each of these sites exhibited a renewed rate of decline not seen in the previous 30 years. By this time, woody plant cover was lower in these sites (13.9% to 17.0%) than in any other portion of the study area. Whereas the rural Lampasas County watersheds displayed a V-shaped trend in woody plant cover and Lampasas Site 9 experienced a steady increase in shrub cover, the Mills County watersheds have undergone a steady, progressive decrease in cover (Figure 2.6). Though this area had much greater levels of woody plant cover at the beginning of the study period, by 2012 it was much lower.

Considering the entire area encompassed by the eight watersheds we investigated, overall woody plant cover trends generally resemble those of watersheds in rural Lampasas County, which make up 72% of the total area. The V-shaped trend is still apparent (Figure 2.7). However, by incorporating trends in Mills County, woody plant cover across the entire area was lower in 2012 (26.5%) than in 1937-1941 (28.4%).

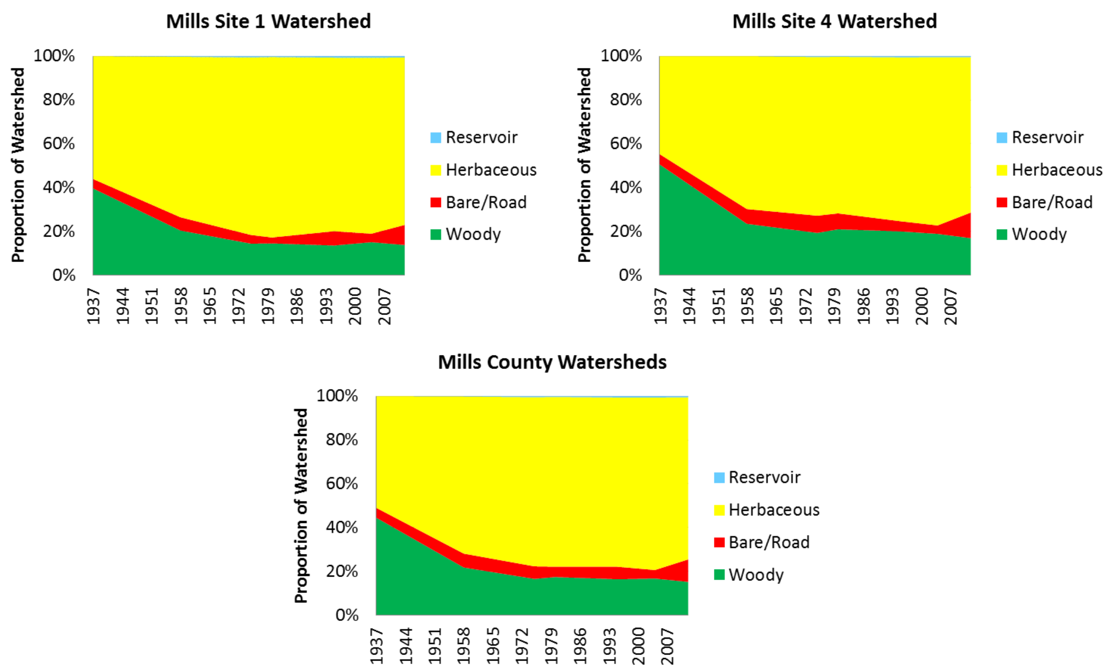


Figure 2.5. Changing land cover over time in Mills County watersheds, each in rural portions of the county. A summary chart combining both watersheds is included. These watersheds have experienced marked decline in woody plant cover, with much slower decreases over the last 50 years.

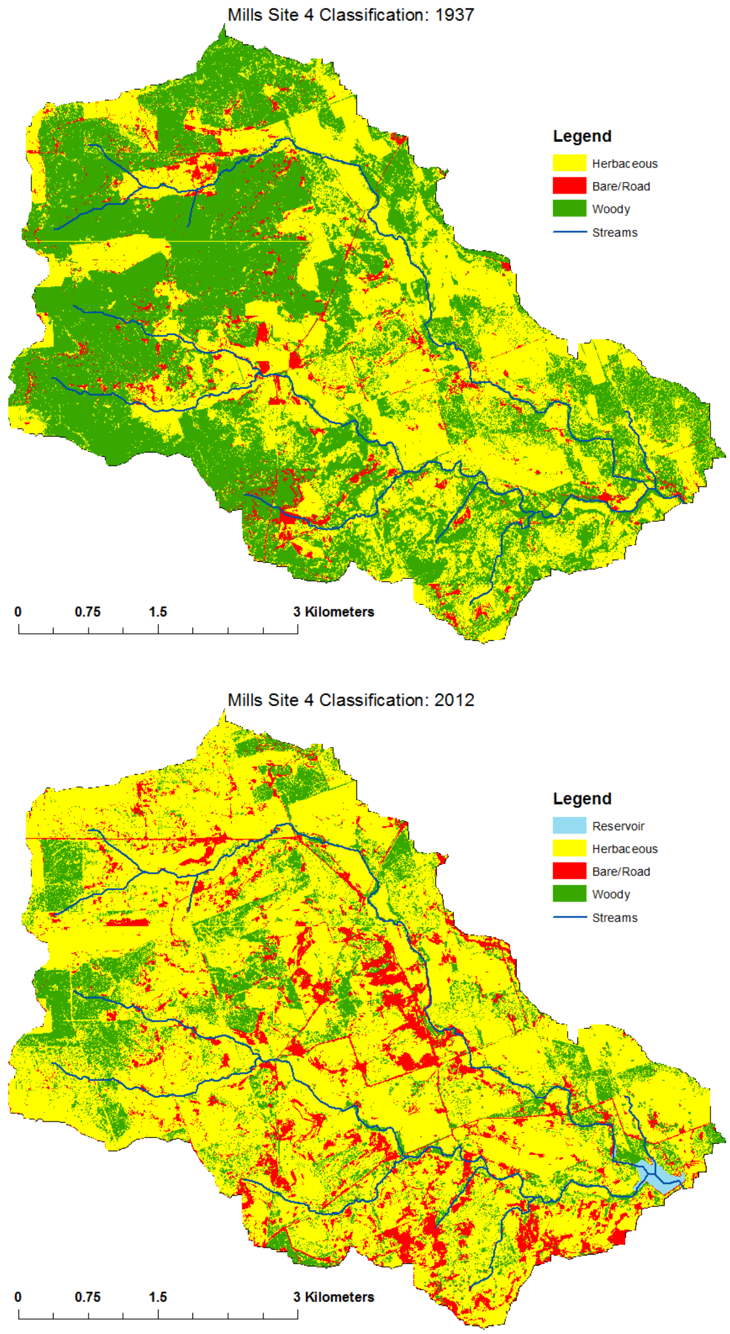


Figure 2.6. Comparison of land cover in Mills Site 4 watershed between 1937 (top) and 2012 (bottom). Woody plant cover has undergone widespread decrease. Increases in red are related to bare soil resulting from intense drought.

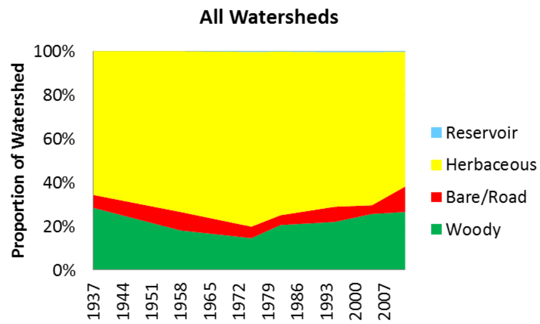


Figure 2.7. Changing land cover over time in all study watersheds, Lampasas and Mills Counties. Total woody plant cover in 2012 was slightly less than in 1937-1940.

Cropland Trends

While woody plant cover has displayed distinct trends in different landscape settings, cropland in all areas has seen consistent precipitous increases, except for localized short-term revivals of a few previously abandoned fields. In the first year of imagery, much more of each individual watershed was in cropland (7.9% to 27.2%) than in 2012 (< 1.0% to 7.3%, Table 2.4). Over this time, the watershed of Lampasas Site 1 lost 60% of its cropland, with Lampasas Site 2 at 98%, Lampasas Site 3 at 82%, Lampasas Site 4 at 92%, Lampasas Site 9 at 70%, Lampasas Site X at 81%, Mills Site 1 at 81%, and Mills Site 4 at 64%. In most cases, the fastest rate of cropland abandonment occurred between 1974/75 and 1980/82 (Figures 2.8 and 2.10). Many watersheds experienced a slight increase in active cropland area in the years preceding 1995, though this was short-lived.

Table 2.4. Cropland by area and proportion of watershed for each watershed over time.

		1937-41	1958	1974-75	1980-82	1995	2004	2012
Lampasas Site 1*	Area (km^2)	6.72	5.76	5.33	4.02	4.49	2.76	2.65
	%	13.2	11.3	10.5	7.9	8.8	5.4	5.2
Lampasas Site 2	Area (km^2)	2.30	2.09	1.05	0.27	0.17	0.25	0.05
	%	9.9	9.0	4.5	1.1	< 1.0	1.1	< 1.0
Lampasas Site 3*	Area (km^2)	6.37	5.79	4.07	0.87	1.97	1.13	1.17
	%	23.4	21.1	14.8	3.2	7.2	4.1	4.2
Lampasas Site 4	Area (km^2)	4.63	2.85	1.85	1.44	0.99	0.46	0.37
	%	11.2	6.9	4.5	3.4	2.4	1.1	< 1.0
Lampasas Site 9	Area (km^2)	0.10	0.02	0.02	0.03	0.05	< 0.01	0.03
	%	7.9	1.9	1.6	2.3	4.3	< 1.0	2.3
Lampasas Site X	Area (km^2)	2.60	2.11	1.73	1.00	1.48	0.55	0.49
	%	11.3	9.1	7.5	4.3	6.4	2.4	2.1
Lampasas Sites	Area (km^2)	22.73	18.63	14.06	7.61	9.15	5.16	4.75
	%	13.6	11.1	8.4	4.6	5.5	3.1	2.8
Mills Site 1	Area (km^2)	9.41	7.63	3.51	2.43	2.96	2.13	1.79
	%	27.2	22.1	10.1	7.0	8.6	6.2	5.2
Mills Site 4	Area (km^2)	5.68	4.56	3.74	2.18	1.44	1.59	2.05
	%	20.3	16.3	13.4	7.8	5.2	5.7	7.3
Mills Sites	Area (km^2)	15.09	12.19	7.25	4.61	4.41	3.72	3.84
	%	24.1	19.5	11.6	7.4	7.0	5.9	6.1
All Sites	Area (km^2)	37.82	30.81	21.31	12.23	13.55	8.88	8.59
	%	16.5	13.4	9.3	5.3	5.9	3.9	3.7

*Due to image availability, very minor portions of coverage from 1940 and 1982 for Lampasas Sites 1 and 3 were omitted. In each case, the omitted area is less than 1% of the total watershed area. For all other years examined, no cropland was located in these areas.

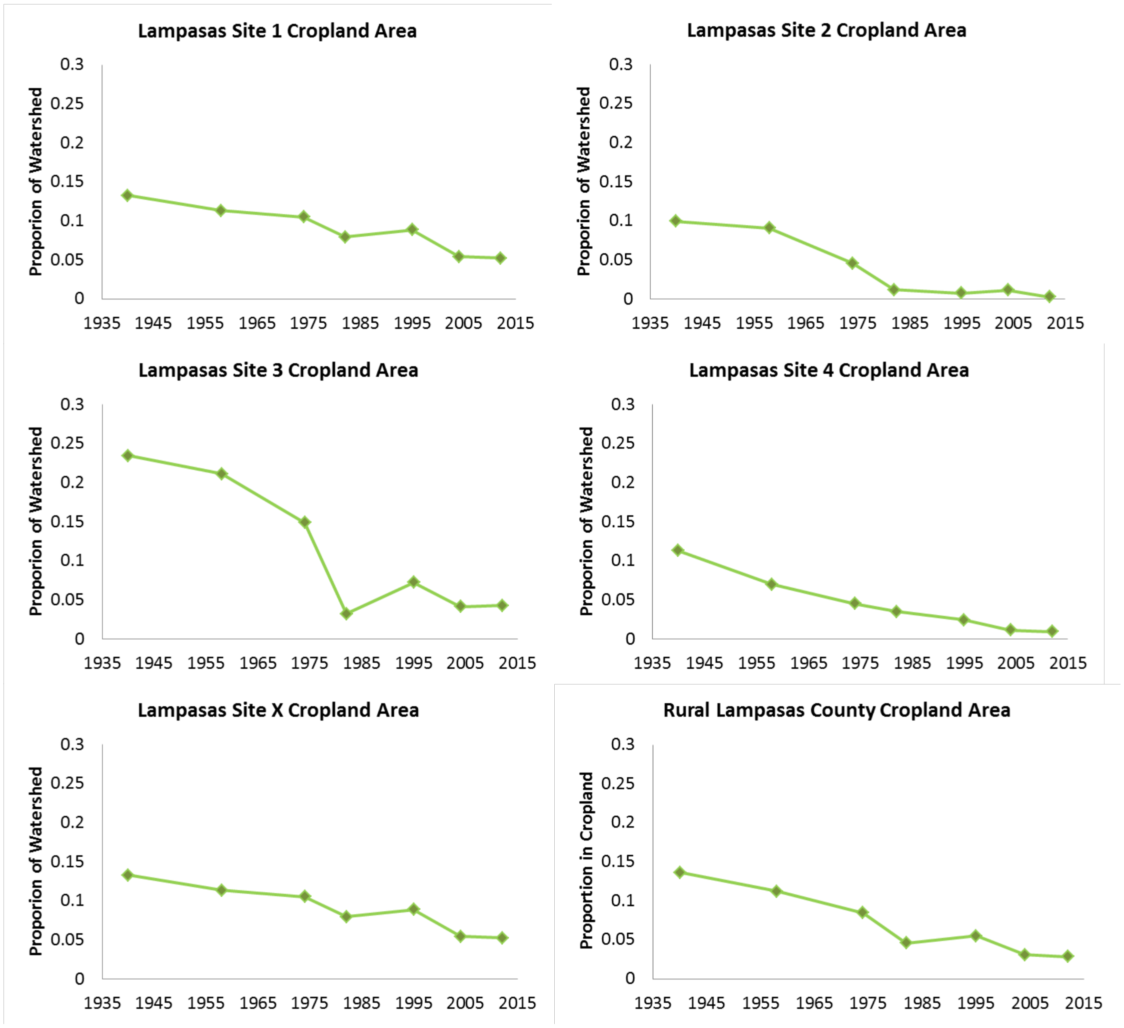


Figure 2.8. Cropland abandonment in rural Lampasas County watersheds. A summary chart combining all rural Lampasas watersheds is included. Steady decline is apparent in each area, with few periods of accelerated decline or recovery.

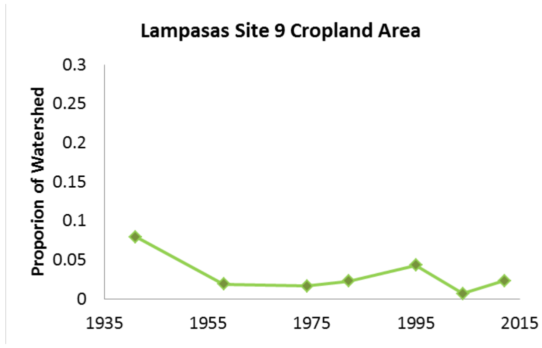


Figure 2.9. Cropland trends over time in peri-urban Lampasas Site 9 watershed, partially within city limits. This area is much smaller and more urbanized than other watersheds.

Cropland trends in the small Lampasas Site 9 watershed are somewhat erratic, but this location had already to undergone some urbanization by 1941, with residential development and a large municipal cemetery in the area. Cropland as a proportion of watershed area was the lowest of any watershed at this time (7.9%, Figure 2.9). Periodic increases in cropland in this area are mostly due to installation of large, periodically mowed green spaces associated with landscaping of the cemetery and surrounded area. Over time, maintenance of some of these areas declined.

Combined, the two watersheds in Mills County experienced a slightly lower degree of cropland loss (75%) than those in Lampasas County (79%), and cropland as a proportion of total watershed area was higher both at the beginning and the end of the period in the Mills County watersheds. As in the rural Lampasas County watersheds, there has been little change since 1980 among croplands in this area (Figure 2.10).

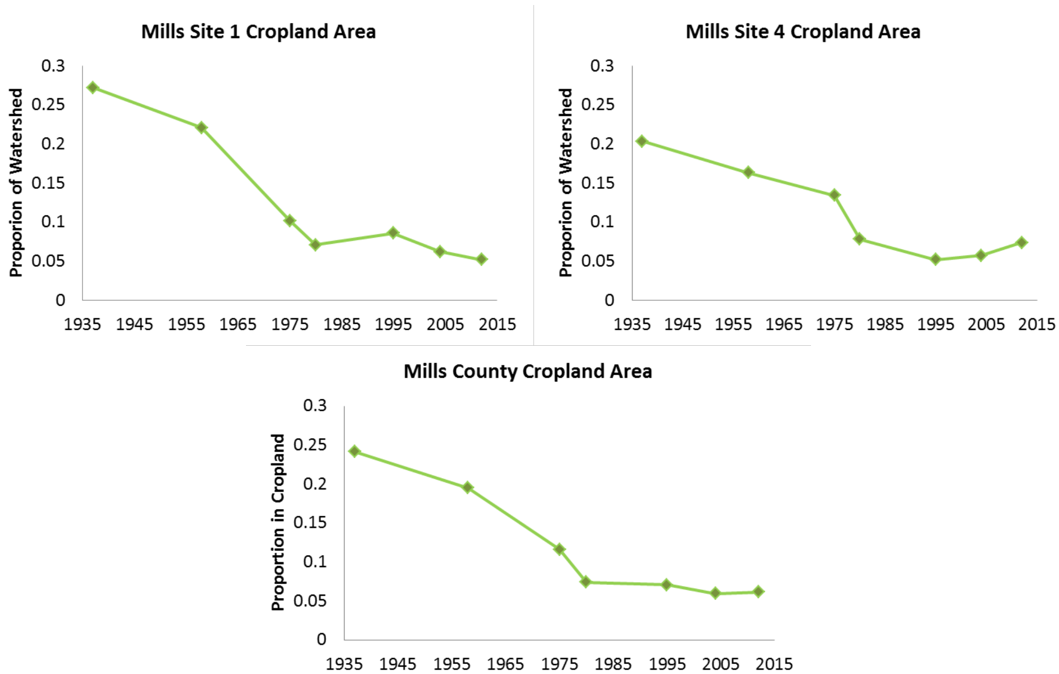


Figure 2.10. Cropland abandonment in Mills County watersheds. A summary chart combining both watersheds is included. Greatest decline occurred in the years prior to 1980, with little change since that time.

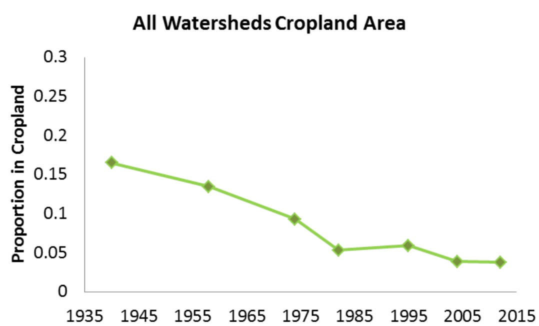


Figure 2.11. Total cropland trends in watersheds in both Lampasas and Mills Counties. The entire area has been marked by a long-running gradual decline in cropland area.

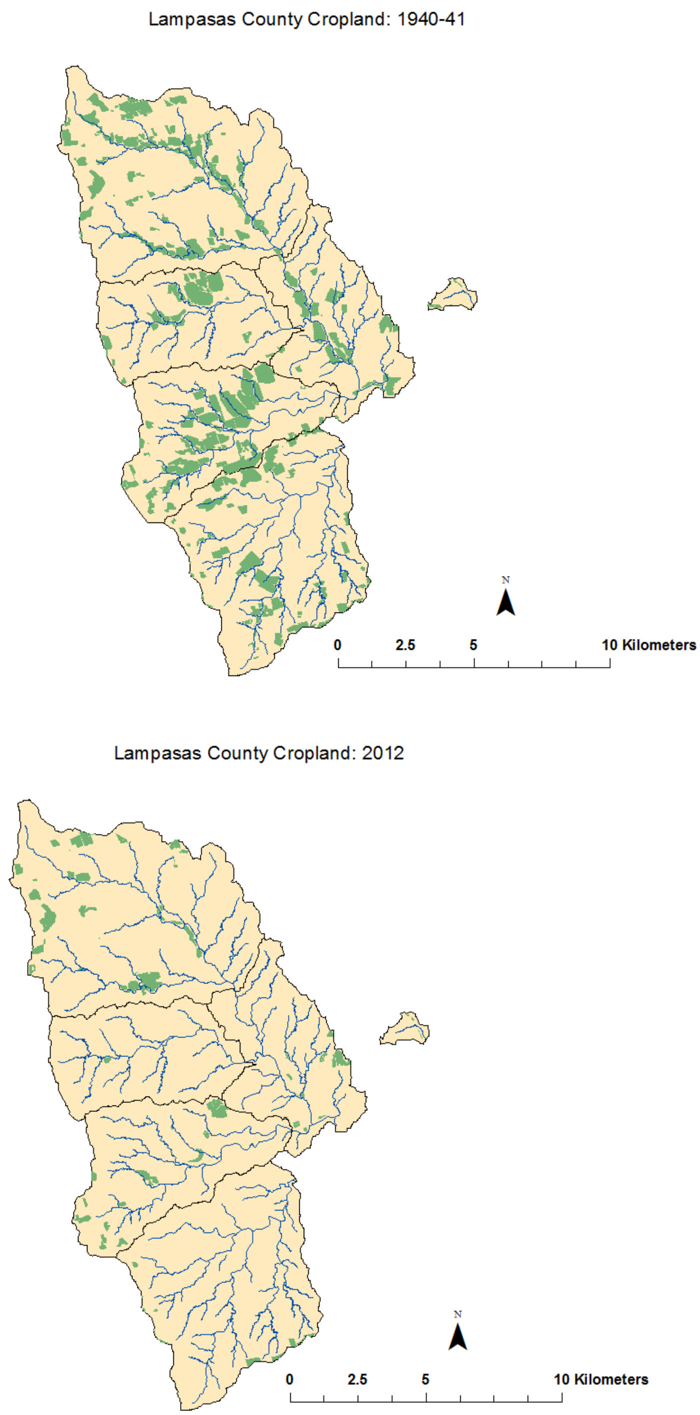


Figure 2.12. Cropland distribution in Lampasas County watersheds in 1940-41 (top) and 2012 (bottom)**.** Cropland coverage has decreased nearly 80% in this area.

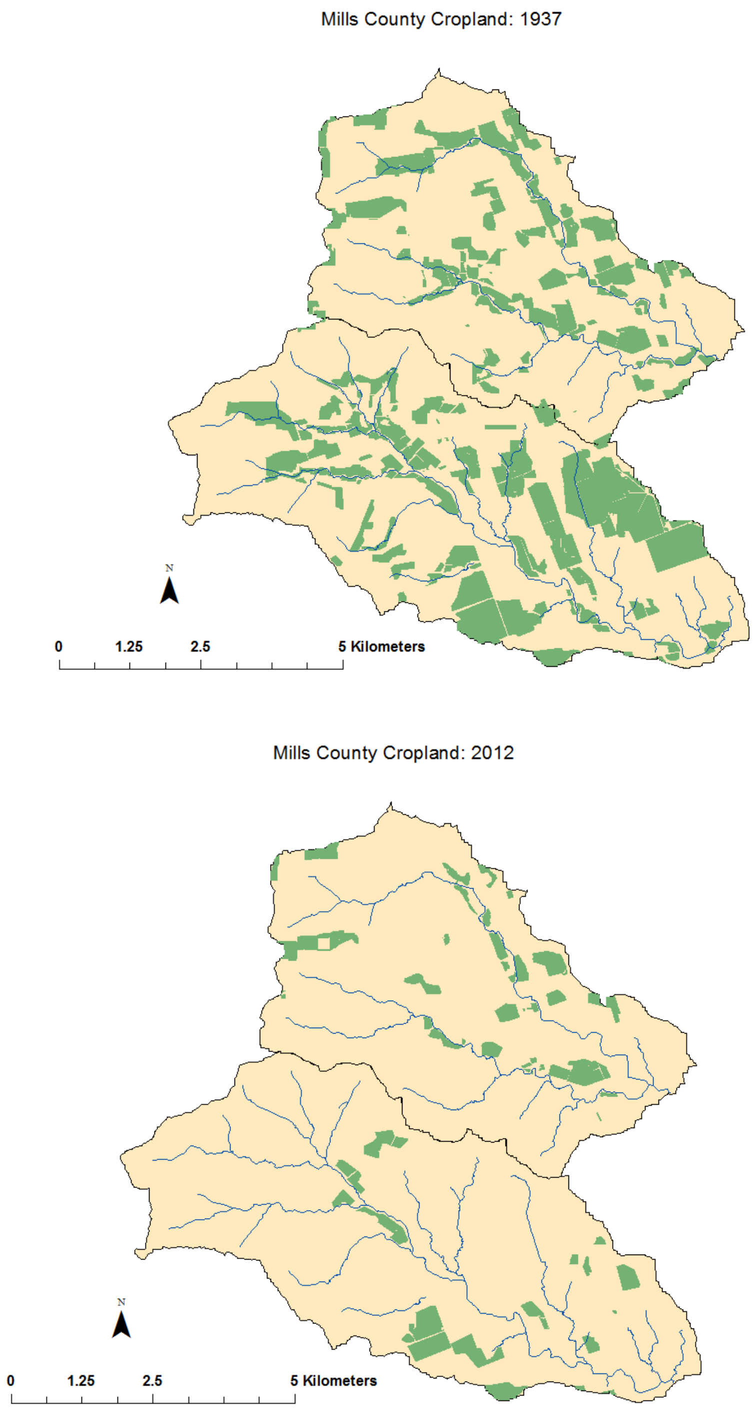


Figure 2.13. Cropland distribution in Mills County watersheds in 1937 and 2012**.**

Cropland coverage has decreased approximately 75% in this area.

Overall, more than 77% of cropland was abandonment across the area between the first year of imagery and 2012. This represents a fall from 16.5% to 3.7% of total watershed area and a loss of nearly 30 km^2 of crop production area over more than 70 years (Figure 2.11). By the end of the study period, croplands were a much smaller part of the regional landscape (Figures 2.12 and 2.13).

Discussion

The trends in woody plant cover seen here suggest much more complicated dynamics at work in central Texas than commonly accepted. Escarpment live oak (*Quercus fusiformis* Small) has been present in the watersheds here since before the beginning of the study period and honey mesquite (*Prosopis glandulosa* Torr.) and other leguminous shrubs are also found at low densities, but vast majority of woody plant cover change has occurred among Ashe juniper. Yet while juniper has indeed increased in some portions of the study area, expansion certainly has not been universal in time or space, even in these eight watersheds. Unexpectedly, this includes widespread decrease in total shrub coverage over portions of the past 75 years, and in some areas this trend continues to the present.

Many smaller-scale studies of woody plant cover on the adjacent Edwards Plateau have documented large increases in juniper cover from the middle of the 19th century (Fuhlendorf, 1992, Smeins & Merrill, 1988). Interestingly, Blomquist (1990) found that the greatest rate of increase in juniper cover occurred between 1973 and 1985, very similar to the highest increase between 1974 and 1982 in parts of the Lampasas Cut

Plain documented in this study. This common trend may be due to a threshold in which regrowth junipers entered a period of the life cycle characterized by rapid growth as well as greater seed production, which further increases recruitment (Smeins & Fuhlendorf, 1997). Parts of the adjacent Edwards Plateau experienced a two- to four-fold increase in woody plant cover between 1949 and 1983, and the greatest increase occurred in ungrazed pastures. Overall, juniper emerged as a stronger dominant over oak among woody plant species after this period (Smeins & Merrill, 1988). A recent investigation of the Balcones Canyonlands National Wildlife Refuge along the Colorado River to south of the present study observed a similar, though much muted, decline and rebound in woody plants over the same timeframe, with woody vegetation cover in 1937 (62%) almost identical to that in 2004 (64%). Additionally, though overall cover changed minimally, localized change marked the study period as numerous patches shifted between woody and herbaceous repeatedly (Murray *et al.*, 2013). However, the authors suggested that mortality from the severe 1950s drought played a significant role in woody plant decrease, through the decline continued well after that period. Though some die-off was observed in imagery during and after prolonged drought in our study area, direct losses appeared to be very minor compared with the effects of brush management.

Several mechanisms have been proposed as drivers of woody plant encroachment, including climatic changes (particularly shifts in magnitude and timing of precipitation), increased grazing and seed transport by livestock, diminished browsing by native wildlife species, changes in wildfire characteristics, elevated concentrations of atmospheric carbon dioxide, and increased nitrogen deposition (Archer *et al.*, 2011).

Woody plant cover increases as a result of the combined effects of growth of existing plants and recruitment of new individuals, and several key adaptations lend a competitive advantage to Ashe juniper in central Texas. The species can establish and survive on nearly all soil types, and prolonged, severe droughts appear to favor its dominance over live oak and other woody species (Smeins & Fuhlendorf, 1997, Twidwell *et al.*, 2013b). In general, drought can be an important driver of mortality in the dynamics of shrub-dominated systems, and extended dry conditions can cause very high levels of woody plant mortality (Laliberte *et al.*, 2004, Twidwell *et al.*, 2013b). Larger adult junipers appear more susceptible to drought than smaller plants, which may affect ecosystem succession and reinforce the establishment of young plants. Additionally, greatest survival of Ashe juniper seedlings occurs beneath established juniper canopy, while greatest growth rates are found among seedlings on the edge of the canopy (Van Auken *et al.*, 2005). This contributes to maintenance and longevity of existing stands while expanding outward at the periphery. Similarly, in a Kansas landscape historically dominated by grassland species, Loehle et al. (1996) identified a threshold of woody plant cover of approximately 20%, beyond which further encroachment occurs more rapidly. However, changes in our study occurred somewhat consistently. In fact, continued expansion of total woody plant cover appeared to level out after the 20% level was exceeded.

In Oklahoma, where it occurs, juniper encroachment has been found to occur preferentially in deciduous woodlands, being relatively uncommon in native grassland (Coppedge *et al.*, 2007). However, this reinforces a transition of even existing woody plant communities to systems that are much more evergreen in character. Over time, fragmentation of grasslands makes even those sites much more susceptible to invasion, and there is much uncertainty regarding the implications of juniper invasion into deciduous woodlands (Coppedge *et al.*, 2001).

In many contexts this conversion is undesirable, and widespread brush management efforts have targeted juniper for removal in favor of grasses or even other woody plants. However, even after intensive management activities on sites where Ashe juniper previously has been established, a viable seed bank remains, ready to initiate regrowth of woody plants (Van Auken *et al.*, 2005). As is the case in all ecosystems, regrowth strategies have a critical effect on ecological trajectories following disturbance, whether natural or anthropogenic. In situations where livestock production is intended, grazing management following brush control efforts is critical to establishing herbaceous plant species. Unless follow-up measures are taken to direct ecosystem function following woody plant management, the primary motivation for most such efforts may be for naught.

One of the greatest concerns over this land cover transition is the threat posed to livestock production, though shrub-animal interactions seem to be quite multifaceted. Differences in forage preference by different livestock greatly affect the early stages of plant community succession following brush management. Goats are responsible for exclusion of oak and juniper saplings and a general decrease in many woody plant species compared with sheep and cattle (Huss, 1954). In an analysis of livestock trends throughout different regions of Texas, Wilcox *et al*. (2012) documented animal numbers in the Lampasas Cut Plain. They found high total numbers in 1940, dominated by sheep and goats. By the mid-1960s, these species had experienced dramatic local decline, and cattle became the dominant grazers in the region, with overall numbers relatively stable over the last few decades. This trend shows remarkable similarity in timing with changes in total woody plant cover in the watersheds of this study, and it is possible that the change in livestock played a role in the cessation of woody plant decline when goat numbers dropped precipitously in favor of cattle.

Though much effort has gone toward identifying the means by which woody plants recruit, grow, and begin to exert dominance in the landscape, it is important to consider that changes in plant cover over large areas are dictated by the sum of recruitment, growth, and mortality. This last component often receives less focus, yet we believe it is the most critical piece of the Ashe juniper puzzle in central Texas. This is highlighted in considering that mortality acts on very short time spans compared with the period required for recruitment and growth of an adult shrub. In addition to the potential effects of livestock grazing that have been suggested as playing a role in the expansion of woody plant cover, the watersheds investigated here have experienced many waves of targeted shrub control efforts over the entire period of study. Various methods of mechanical removal have been widespread since the earliest years of settlement, ranging from simple means such as grubbing with hand tools to the utilization of motorized heavy machinery (Hamilton & Hanselka, 2004). Even when total woody plant cover is relatively stable, patches and individual canopies can be quite dynamic under natural conditions (Browning *et al.*, 2010). Results from the Mills County watersheds suggest that this effect also can be achieved through ongoing brush management efforts. Classification results for individual years in a given watershed indicate that internal dynamics are at work even when overall woody plant cover shows limited change. In many cases, woody plant expansion in certain areas is balanced by reduction in others within the watershed, and the rate of turnover is quite high. In addition to other efforts to remove woody plant cover in the area, the Bracero Program resulting from the Mexican Farm Labor Program Agreement of 1942 supported the use of migrant labor across southern portions of the United States through a temporary visa program. In Texas, much of this labor was directed at juniper control in central Texas, including the drainages examined in this study (Conner, 2013, Garriga *et al.*, 1997). Curiously, the period of woody plant decline in the examined watersheds closely parallels the period over which the labor program operated (1942-1964). Small-scale brush management, even at the level of an individual field, can profoundly change the character of a regional ecosystem, particularly as effects are summed across a wide area. Indeed, cycles of clearing efforts have been responsible for rapid fluctuations in woody plant cover elsewhere in Texas. Even on the largest of scales, across the Great Plains, cyclical brush clearing efforts are one of the most extensive aspects of landscape change over the last four decades, behind only widespread enrollment in conservation programs and urban development (Drummond *et al.*, 2012). Where the application of land management practices occurs with sufficient frequency, Coppedge et al. (2001) suggested that even over very long periods, some landscapes may be virtually immune from woody plant encroachment due to consistent, intensive management.

While livestock grazing has critical place in the history of the Lampasas Cut Plain and mechanical brush management has played a dominant role in local woody plant dynamics over the past century, we believe that something greater has driven the perceived change in vegetation characteristics. Though grazing and woody plant management efforts have been significant and widespread over the last century and exercise great influence at plot and field scales, in all likelihood the removal of periodic fire from the ecosystem is an even more important overall driver in this landscape. In other settings across the southern Great Plains, the conversion of prairie to woodland has been linked to long-term fire suppression efforts having significant impacts on species distribution at larger scales (Fuhlendorf & Smeins, 1997, Huss, 1954, Twidwell *et al.*, 2013a).

In identifying the proximal causes of landscape change, we must understand the extent of the change that has occurred. Considering a historical reference point is important to interpreting woody plant dynamics as encroachment or decrease (Browning *et al.*, 2009, Murray *et al.*, 2013). Despite being frequently considered an invasive plant species, Ashe juniper, along with more socially accepted live oak, are more correctly the successional climax of many areas that have appeared as prairies at certain points in history (Nelle, 1997b). Even at the turn of the 20th century, the Edwards Plateau and adjoining regions were described as being rich with juniper and oak woodland resources to meet the needs of growing communities (Bray, 1904). While there were concerns of localized overexploitation, even at that time it was observed that expansion of woody plant cover more than compensated for increasing harvest. Trees and shrubs have been a significant part of the southern Great Plains since the earliest historical accounts. Huss (1959) suggested that woody plant communities in the region actually are quite stable, having developed under somewhat constant edaphic, climatic, and geological conditions that are generally unchanged in the region. Previous fire regimes provided a regular level of disturbance that opened woodlands and supported an abundance of herbaceous vegetation, but this variable has largely remained absent since the first Europeans arrived in central Texas. In many landscapes dominated by shrubs, encroachment often continues until a dynamic equilibrium is achieved. Studies have identified underlying soil type and mean annual precipitation as key factors in determining encroachment rates and total woody plant cover in a climax state (Browning *et al.*, 2008, Laliberte *et al.*, 2004). With the removal of fire from the landscape, this level of dynamics equilibrium has shifted, though the potential for widespread woody vegetation remains high.

Woody plant encroachment is one piece of dynamic land use and land cover change in the southern Great Plains. Another key piece of this complex regional system is that of cultivated agricultural lands. One commonly cited possible reason for the increase in juniper and other woody species is the observation that greater time elapsed since cropland abandonment generally leads to dominance by shrub or tree communities (Benjamin *et al.*, 2005) in old fields. Conceptually, this is certainly a potential candidate for explaining the recent increase in juniper and other species in parts of the Lampasas Cut Plain. A great deal of formerly cultivated land has been released to rangeland use, and this has been seen in many other areas in the state, though land use changes are largely site-specific.

The Lampasas Cut Plain has had a well-recorded yet complex history of agricultural production. At the national scale, cropland conversion began in the eastern United States and swept through the Great Plains to the west, accompanying expansion of European settlement and population growth. Already by 1860, parts of the Lampasas Cut Plain had become important in regional wheat production (Jordan-Bychkov *et al.*, 1984). In the past, land was cleared and abandoned, often concurrently and in close proximity. Total farmland area in the United States showed steady increases until 1950, when overall cropped area began to decrease in many areas, particularly in the East (Hart, 1968). In eastern regions of the country with long agricultural histories, peak farm acreage occurred prior to 1910, while others took much longer to reach their maximum. Cropland in much of the southern Great Plains reached its peak between 1910 and 1930 (Ramankutty & Foley, 1999b). Elsewhere on the Great Plains, cropland cultivation reached a peak by the 1930s before undergoing a series of widespread declines due to severe droughts (Coppedge *et al.*, 2001). However, rather than being abandoned long-term as in our portion of the Lampasas Cut Plain, production often returned when weather conditions were more favorable. There the grassland-cropland mosaic underwent frequent shifts largely driven by short-term socioeconomic factors.

Abandonment has followed the same general geographical pattern as initial attempts at cultivation, sweeping from east to west across North America. There appears to be a temporal lag in cropland abandonment related to the initial time the land was converted to agricultural use (Hobbs & Cramer, 2007, Ramankutty & Foley, 1999b), and the majority of cropland abandonment has occurred in areas previously covered by forests and woodlands. Between 1930 and 1970, every county in the eastern two thirds of Texas lost more than 30% of its cropland area, with many experiencing abandonment rates of more than 65% (Roberts, 1987). Even at the scale of global analysis of cropland changes since 1700, the abandonment of cultivated lands in a small part of the southern Great Plains occupied by the Lampasas Cut Plain is visible over the period 1940-1960 (Ramankutty & Foley, 1999a). Farmland abandonment continued consistently and was widespread across the eastern United States between 1973 and 2000 (Loveland & Acevedo, 2006). However, in the Great Plains in eastern Colorado, agricultural land increases occurred well into the 1960s and then again in the 1970s, slowed temporarily only by severe drought (Parton *et al.*, 2003). Where recent increases in cropland have occurred, they frequently are associated with irrigated agriculture (Parton *et al.*, 2003). This mode of production is very uncommon in the Lampasas Cut Plain, and local increases are practically nonexistent.

Enrollment of Lampasas Cut Plain agricultural lands in the Conservation Reserve Program (CRP) was very high by 1960 (Hart, 1968), and regional participation among marginal lands across much of the country has remained high to the present (Loveland & Acevedo, 2006). By 1980, nearly all of the wheat production that dominated early agricultural activity had shifted to the east (Jordan-Bychkov *et al.*, 1984). Across the entire Great Plains, much additional cropland was abandoned in the 1980s and 1990s and enrolled in these conservation programs (Drummond *et al.*, 2012).

The seeds of farm abandonment in the southern Great Plains and elsewhere were sown in the initial stages of land conversion, which were accompanied by unrealistic expectations of land productivity and sustainability, as well as improper management and conservation practices (Booth, 1941, Parton *et al.*, 2003). Average annual precipitation in the Lampasas Cut Plain generally is considered sufficient to sustain the major local crops of wheat, oats, and different varieties of sorghum (Allison, 1991, Clower, 1980). However, "average" conditions are rare indeed, and periodic droughts have presented significant hardships to local farmers at many points in the past. Counties immediately to the west long have been marked by agricultural activities that strongly favor livestock production (Bentley, 1898), indicating that the Lampasas Cut Plain lies on or very near a climatic threshold delineating the viability of sustainable crop production.

Agricultural lands are part of a complex system of interactions between sociological and ecological factors, and reasons for abandonment may involve anything from declining soil fertility to rising land prices for conversion nonagricultural uses (Hobbs & Cramer, 2007). In most cases, cropland loss is strongly associated with population loss (Roberts, 1987). Cropland abandonment has been connected to poor soils, drought, socioeconomic conditions, urban expansion, and conservation policies, with precise drivers varying from region to region (Drummond, 2007, Hart, 1968, Jakubauskas *et al.*, 2002). Overall, the limitations imposed by the physical environment represent the strongest determinant of continued agricultural production. At larger scales, cropland abandonment has occurred due to competition with production in other areas as well as increased competition for other land uses. When the loss of a major crop changes local agricultural activities, abandonment can be avoided if an alternative enterprise is economically and environmentally feasible (Hart, 1968). In Lampasas County and Mills County, these alternatives have not been different crops but rather different land uses, including the increase in cattle identified by Wilcox et al. (2012). Abandonment often includes a long, gradual period of consistent underuse, marked by declining intensity and extent. Significant proportions of farm area may not be planted or even harvested in early stages due to anticipated climatic conditions and economic considerations (Michaels, 1985). As a result, effective farm acreage may be greatly diminished long before a change is reflected in agricultural census data (Hart, 1968). Almost universally these areas gradually revert to prairie, shrubland, and woodland. In areas where both rangelands and croplands are present, cultivated lands tend to be located on alluvial soils (Burke *et al.*, 1994). This is true of the watersheds examined here, particularly as cropland distribution is followed through time. Cultivated areas in uplands have decreased over the last several decades.

Long-term trends in cropland area are dynamic and can be reversible in many landscape settings (Jakubauskas *et al.*, 2002). Marginal lands have the highest rates of land use change and may alternate between rangeland use and dryland farming, depending on market conditions and federal support (Drummond *et al.*, 2012). Indeed, this may explain the trend among small portions of cropland area in the examined watersheds over the last couple decades, as certain fields fallowed for long periods briefly were cultivated once again. However, that generally has not been the case in our study area and declines have been consistent and extensive. The plains of our study site are part of a larger ecoregion that has some of the lowest rates of persistent agriculture in all of the Great Plains. In fact, regions dominated by shrubs and woodlands that require frequent brush management are marked by low levels of persistent crop production. To keep pace with this woody plant regrowth and maintain land for various agricultural activities, significant forest clearing has continued across the southern Great Plains, even accelerating in recent decades (Ramankutty *et al.*, 2010). As agriculture becomes impractical in some areas, it is sometimes shifted to others. Cropland loss due to urbanization also has been negligible in this area, aside from some conversion of the Lampasas Site 9 watershed that includes a small portion of the city limits of Lampasas and surrounding area. Regardless of the local drivers of cropland abandonment near the Lampasas River, there is an apparent disconnect between the mostly linear loss of cultivated area and changes in woody plant cover. Indeed, land use and land cover change are not necessarily coupled and occur at different rates (Drummond *et al.*, 2012).

Many croplands in the eastern United States have undergone succession to the original plant communities of woodland and forest, similar to the trend seen in central Texas. Indeed, succession of old fields often results in plant communities that are somewhat characteristic of undisturbed vegetation in the region, if soils are not so degraded as to prevent this restoration (Booth, 1941, Ramankutty & Foley, 1999a). Even after extended periods, some abandoned fields may be very different from the previous undisturbed vegetation. Continued disturbance in the form of heavy grazing or burning of abandoned croplands might hinder development of climax prairie vegetation. Booth (1941) examined a field that not been cultivated for more than thirty years yet still had not undergone succession to a fully-developed prairie. Yet the species dominating in the succession of abandoned fields can be a more important determinant of ecosystem function than the presence or absence of prior cultivation itself (Christian & Wilson, 1999, Knops & Tilman, 2000). A study of old fields in Kansas found productivity of grasslands on former cropland to be greater than that of original grasslands themselves, and ecosystem function resembled that of native prairie after 12 years of abandonment and subsequent restoration efforts (Baer *et al.*, 2002). But in some cases, productivity of restored rangeland may be realized only when seeding is conducted (Bement *et al.*, 1965).

The ecological implications of cropland abandonment may be considered along two major perspectives: that of the potential for restoration to some prior or natural ecosystem or that of the potential loss of certain ecosystem components that depended on agriculture and related activities for continued existence (Hobbs & Cramer, 2007). This appears to be a critical consideration for the Lampasas Cut Plain, where active brush management and passive woody plant encroachment intersect. In many ways, the significance of changes in woody plant cover is interpreted based on how one interprets these landscape changes, whether an emerging recent concern or a long-tenured ecological condition.

Though abandonment of cropland certainly has allowed the establishment of woody plants in some former cultivated fields, questions remain. Total woody plant cover declined in almost all watersheds during the initial stages of cropland abandonment. Additionally, though major loss of cropland area has occurred across the Lampasas Cut Plain, juniper and other woody species have not increased in large portions of the region. The varying trends in different parts of the landscape and the puzzling paired decline of woody plant area and cultivated area prior to 1970 indicate other drivers likely play an important role in the complex dynamics of Ashe juniper and other woody species. As brush management efforts likely will continue into the future due to a variety of motivations, there is great uncertainty over the ecological effects of this combined land use and land cover conversion.

CHAPTER III

RETURN OF THE NOVEL ECOSYSTEM: HISTORICAL CYCLES OF WOODY PLANT ENCROACHMENT FOLLOW HUMAN POPULATION DEMOGRAPHICS AND PRIORITIES

Introduction

Shrub and tree invasion of grasslands has occurred in ecosystems around the globe, with a variety of species and effects (Archer, 1994). Precise causes of this phenomenon are difficult to discern, but several mechanisms have been proposed to explain woody plant encroachment. All of these typically include at least one of changes in fire regime, land use, or climate, and these factors often interact with one another (Archer *et al.*, 1995, Grover & Musick, 1990, Van Auken, 2000). However, most investigations into the causes of these major ecosystem changes have focused on the relationship of plant cover to environmental characteristics or the influence livestock and wildlife.

Woody plant encroachment as a recent trend has not been universal, as time series photography has documented many cases in which trees and shrubs have been key members of the landscape for nearly two centuries, indicating the historical presence of mixed woodland-grassland systems (Archer *et al.*, 2011). Recent analyses of vegetation cover in central Texas indicate trends in woody plants are due to other factors in addition to landscape characteristics of physiography, soils, climate, and livestock grazing typically cited. Rangeland watersheds with similar characteristics in the Lampasas Cut

Plain of central Texas have displayed differing responses in shrub and tree cover over the last several decades. Three pronounced trends in woody plant cover emerged based on landscape setting from the land cover classification analysis in Chapter 1: 1) rural Lampasas County, 2) within Lampasas city limits, and 3) rural Mills County. While the magnitude of trend varies slightly between watersheds in each of these areas, the consistency within each of these clusters is very apparent. We sought to explore the potential relationship between woody plant expansion and management with human population dynamics to explain changes over a period stretching across nine decades.

The objectives of our work are to quantify the timing and magnitude of local changes in 1) woody plant cover and 2) human population in central Texas in addition to 3) identifying the relationship between these variables to explore a potential mechanism for differing trajectories of shrub cover. Given regional anecdotal evidence, we expect to see a steadily increasing trend in both total cover by trees and shrubs as well as in human population. Recent observations indicate these are somewhat closely correlated.

Materials and Methods

Location Description

We examined clusters of eight different watersheds in central Texas. The study area lies in the Lampasas Cut Plain, a transitional zone between the Cross Timbers, the prairies of north Texas, and the Edwards Plateau (Allison, 1991, Clower, 1980). The landscape consists of broad valleys and low mesas and is characterized by a complex mosaic of woody plants and herbaceous vegetation. Most of the region is rural in nature, particularly to the north and west. Watersheds in Lampasas County and Burnet County are approximately 100 km from Austin, Texas and those in Mills County 150 km away. The Mills County watersheds are located just to the east of Goldthwaite (2010 U.S. Census population: 1,878), with Lampasas County sites partially within and immediately to the west of Lampasas (2010 U.S. Census population: 6,681). Land use outside of city limits generally is dominated by rangeland for livestock production, with cropland a much smaller percentage of the area. Median watershed area is 27.74 km^2, with the peri-urban Lampasas Site 9 much smaller (1.24 km^2) than the remaining watersheds. Average annual precipitation ranges from 710 mm in the north to 780 mm in southernmost watersheds. The context of the study watersheds within the landscape is displayed in Figure 3.1.

Imagery Sources

We obtained aerial and satellite imagery from the earliest period available followed by approximately 15-year increments to the present. Images were provided by a number of sources, including the USDA-Aerial Photography Field Office (APFO), Texas Natural Resource Information System (TNRIS), National Agriculture Imagery Program (NAIP), and the private firm, Tobin International, Ltd. Due to varying availability for different portions of the study area, this resulted in imagery from 1937/1940/1941, 1958, 1974/1975, 1980/1982, 1995, 2004, and 2012 (Table 3.1). Late fall and winter images were preferred to take advantage of the contrast between predominantly evergreen woody plants and herbaceous species. Spatial resolution ranged from 1:1667 to 1:40000 and represented a combination of black and white, color infrared, and natural color imagery. Where necessary, hard copies were digitally scanned, and all images were radiometrically corrected and georectified to 2004 NAIP imagery (<1m error). Following the procedure described by Laliberte *et al*. (2004). we then applied a 3 x 3 kernel low-pass filter to reduce spatial frequency. Finally, we resampled all images to a common 1 m resolution using nearest neighbor methodology to standardize analysis and maintain consistency in appearance of landscape features across resolutions from different images.

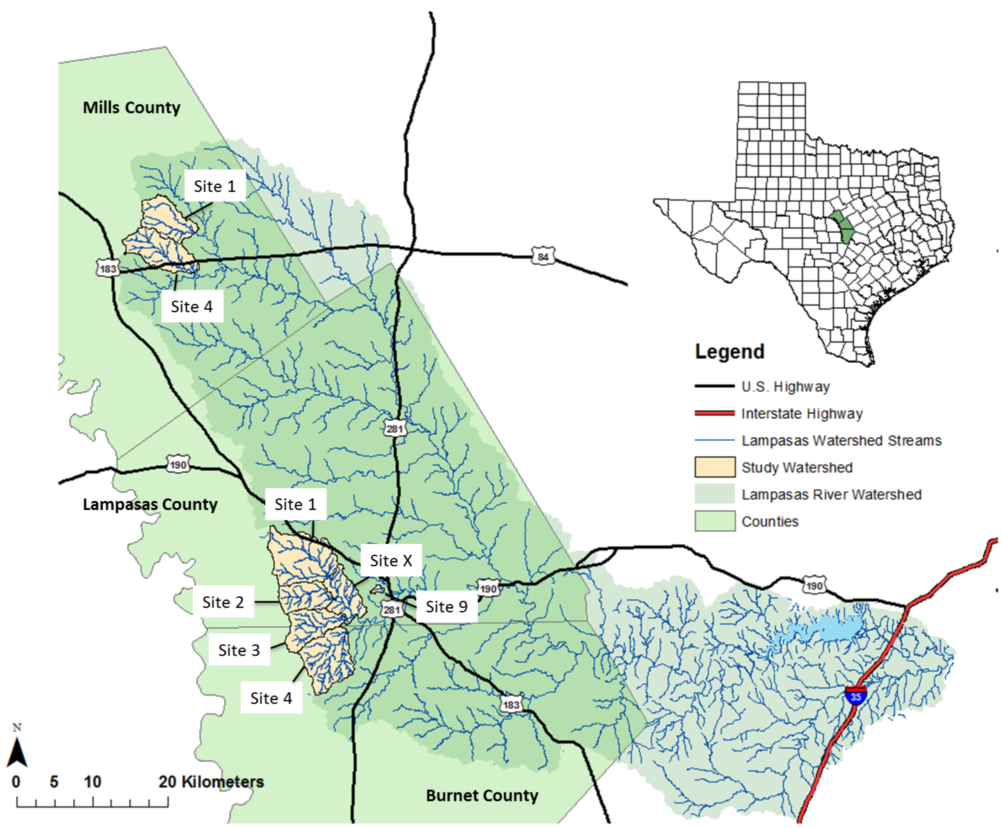


Figure 3.1. Map of study location with highway infrastructure in central Texas.

Table 3.1. Remote sensing imagery characteristics for each image year.

Area	Image Year	Acquisition Date	Type	Spatial Scale	Spatial Resolution (m)
Lampasas Sites 1-4, X	1940	December 1939-August 1941	BW	1:1667	0.45
	1958	December 1957-January 1958	BW	1:20000	0.24-0.51
	1974	February 1974	BW	1:40000-1:85000	1.01-1.02
	1982	December 1982	BW	1:4700	0.90
	1995	January 1995-February 1995	CIR	1:40000	1.01
	2004	September 2004-December 2004	CIR	1:40000	1.00
	2012	October 2012	NC	1:12000	1.00
Lampasas Site 9	1941	August 1941	BW	1:1667	0.45
	1958	December 1957	BW	1:20000	0.24
	1974	February 1974	BW	1:85000	1.02
	1982	December 1982	BW	1:4700	0.90
	1995	February 1995	CIR	1:40000	1.01
	2004	December 2004	CIR	1:40000	1.00
	2012	October 2012	NC	1:12000	1.00
Mills Sites 1,4	1937	December 1937-Novermber 1938	BW	1:1667	0.40
	1958	December 1957-April 1959	BW	1:20000	0.51
	1975	November 1975	BW	1:60000	1.02
	1980	November 1980	BW	1:2640	0.63
	1995	January 1995	CIR	1:40000	1.01
	2004	December 2004	CIR	1:40000	1.00
	2012	October 2012	NC	1:12000	1.00

Woody Plant Coverage

Using the imagery above, we performed image classification using the Example-Based Feature Extraction module with the SVM algorithm in the ENVI 5.0 software package. This object-based approach is advantageous in that it segments imagery into pixel groups based on similar spectral, textural, and spatial characteristics, each with ecological meaning that can be interpreted with expert knowledge (Laliberte *et al.*, 2004, Platt & Schoennagel, 2009). We analyzed imagery for each of the eight watersheds from each year in the time series, with image segments designated as one of three classes: 1) herbaceous cover, 2) bare ground/road cover, and 3) woody plant cover. All objects were assigned to one of these categories. Bare ground and roadways were combined to reflect their similar characteristics, as the majority local rural roadways are unpaved and function much the same as bare ground. Woody plants included each evergreen coniferous, evergreen broadleaf, and deciduous broadleaf trees and shrubs, often identifying individual plants. While imagery from more recent years supports the separation of these groups, older imagery does not, due to spectral and seasonal coverage limitations (Browning *et al.*, 2009).

Population Analysis

To improve our understanding of the relationship between landscape characteristics and factors of human influence on the Lampasas Cut Plain landscape, we obtained historical population data from the 1930, 1950, 1980, 1990, and 2010 editions of the U.S. Census. We plotted population in selected portions of the study area to observe overall trends and then compared woody plant cover against human population over time in each area to document the correlation between these trends. Watersheds in rural Lampasas County, within Lampasas city limits, and in rural Mills County were considered separately, with each cluster compared to human population data for that landscape setting. The Lampasas Sites 1-4 and Lampasas Site X, all in rural portions of Lampasas County, were plotted in terms of the rural population of that county as defined by designations of the U.S. Census data. For Lampasas Site 9, we examined the population of the City of Lampasas, as this watershed lies partially within city limits to the southeast. We determined the social trends for Mills Sites 1 and 4 in rural settings to the north by examining the Mills County population, all of which is considered rural.

In most cases, quality remote sensing imagery was not available for the specific years in which a census was conducted. For those years that do not coincide with decennial census reporting, population estimates for imagery years were made using an arithmetic interpolation approach. For instance, the Mills County population in 1958 was considered to be 0.80 of the difference between that for 1950 and 1960. We used official population estimates for 2012 data. Once population trends were plotted over time, we conducted correlation analysis with woody plant cover to identify the relationship between these variables.

Results

In each landscape setting, trends in human population and woody plant cover show remarkable similarity. In the rural Lampasas County watersheds, human population declined from a high in 1940 to a minimum in 1970 before rising once again to reach maximum levels at the present day (Figure 3.2). The same behavior is displayed among woody plant cover in the area over the same period, with even periods of more rapid increase nearly identical. Population in the City of Lampasas has increased consistently since 1940, with growth slowing considerably in recent years (Figure 3.3). Total woody plant cover in the watershed of Lampasas Site 9 shows a somewhat similar pattern, with gradual increases since 1941 and relatively stable levels in the last 30 years. Mills County population has decreased dramatically since its historical maximum in 1930, with some minor fluctuations at a much lower level over the last few decades (Figure 3.4). This same trend holds for woody plant cover in the Mills County watersheds. Rural populations in both counties operated on similar scales, though exhibiting different trends. Naturally, the population of Lampasas was higher than the surrounding areas.

When woody plant cover is plotted against population in each area, the apparent similarities are borne out, with the relationship much more visible. Rural Lampasas County woody plant cover is closely correlated with Lampasas County rural population with $r^2 = 0.88$. Likewise, woody plant cover in the peri-urban Lampasas Site 9 watershed is correlated with City of Lampasas population with $r^2 = 0.82$. Finally, woody plant cover in the rural Mills County watersheds was correlated with Mills County total population with $r^2 = 0.93$.

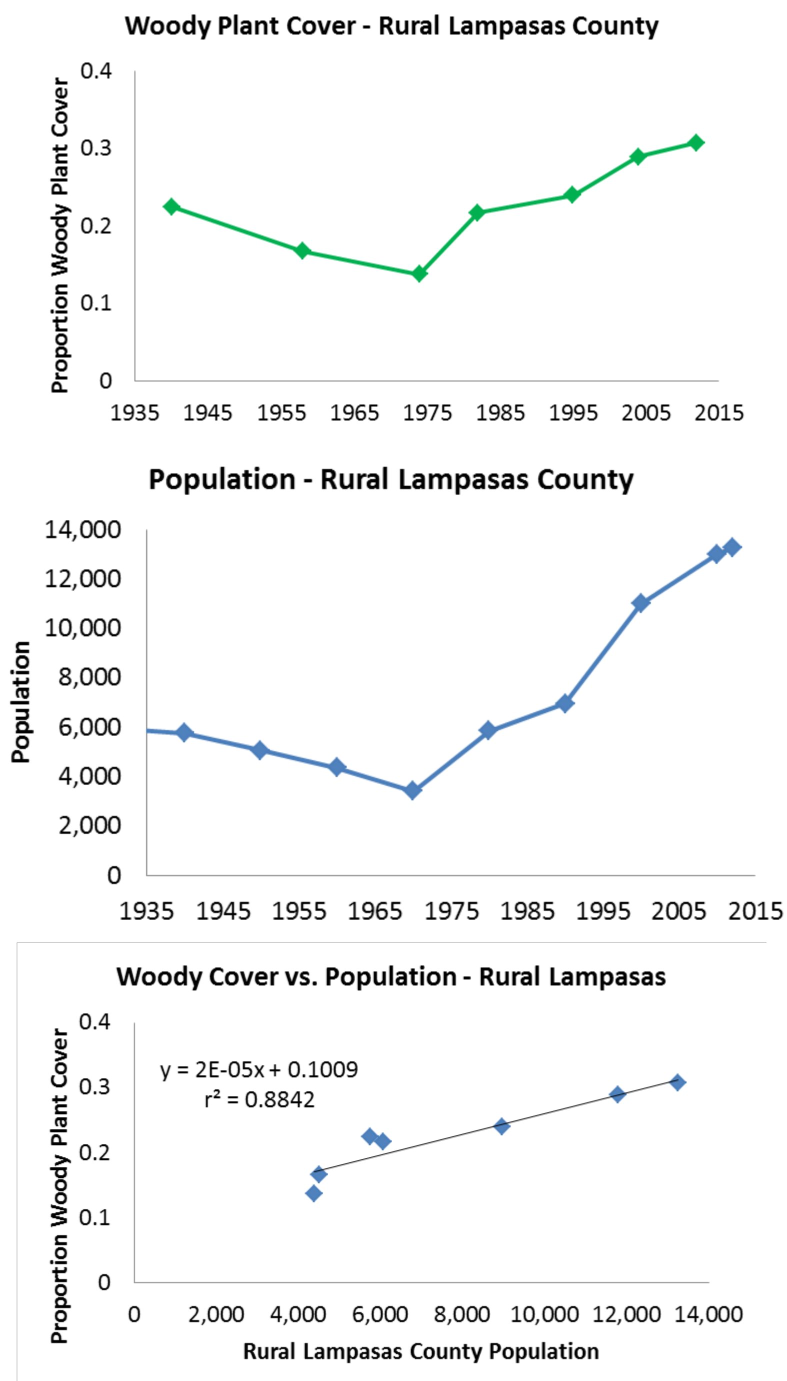


Figure 3.2. Trends in woody plant cover (top) and rural population (middle) for watersheds in rural Lampasas County follow the same pattern and are closely correlated (bottom).

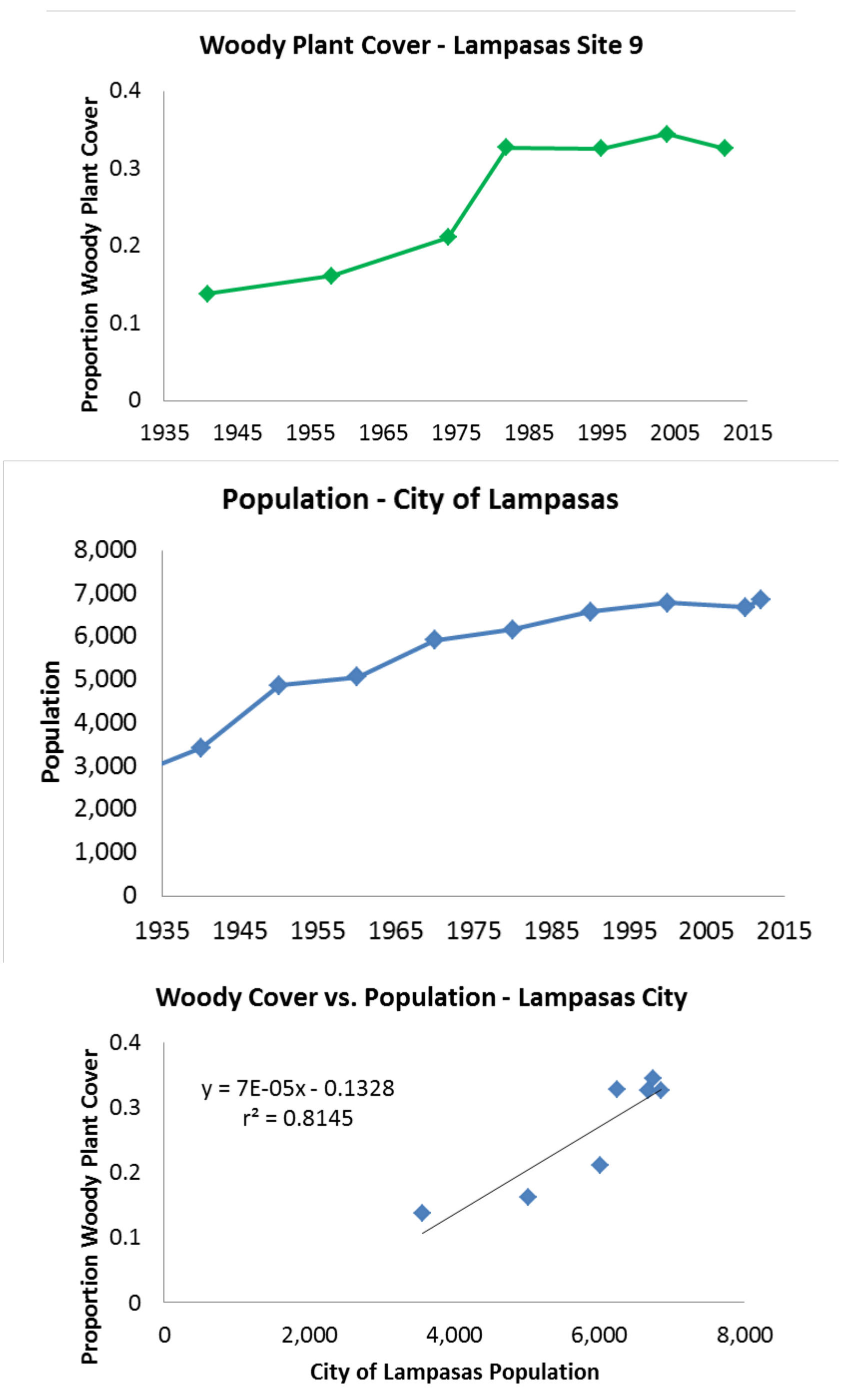


Figure 3.3. Trends in woody plant cover (top) and City of Lampasas population (middle) for peri-urban Site 9 watershed follow the same pattern and are closely correlated (bottom).

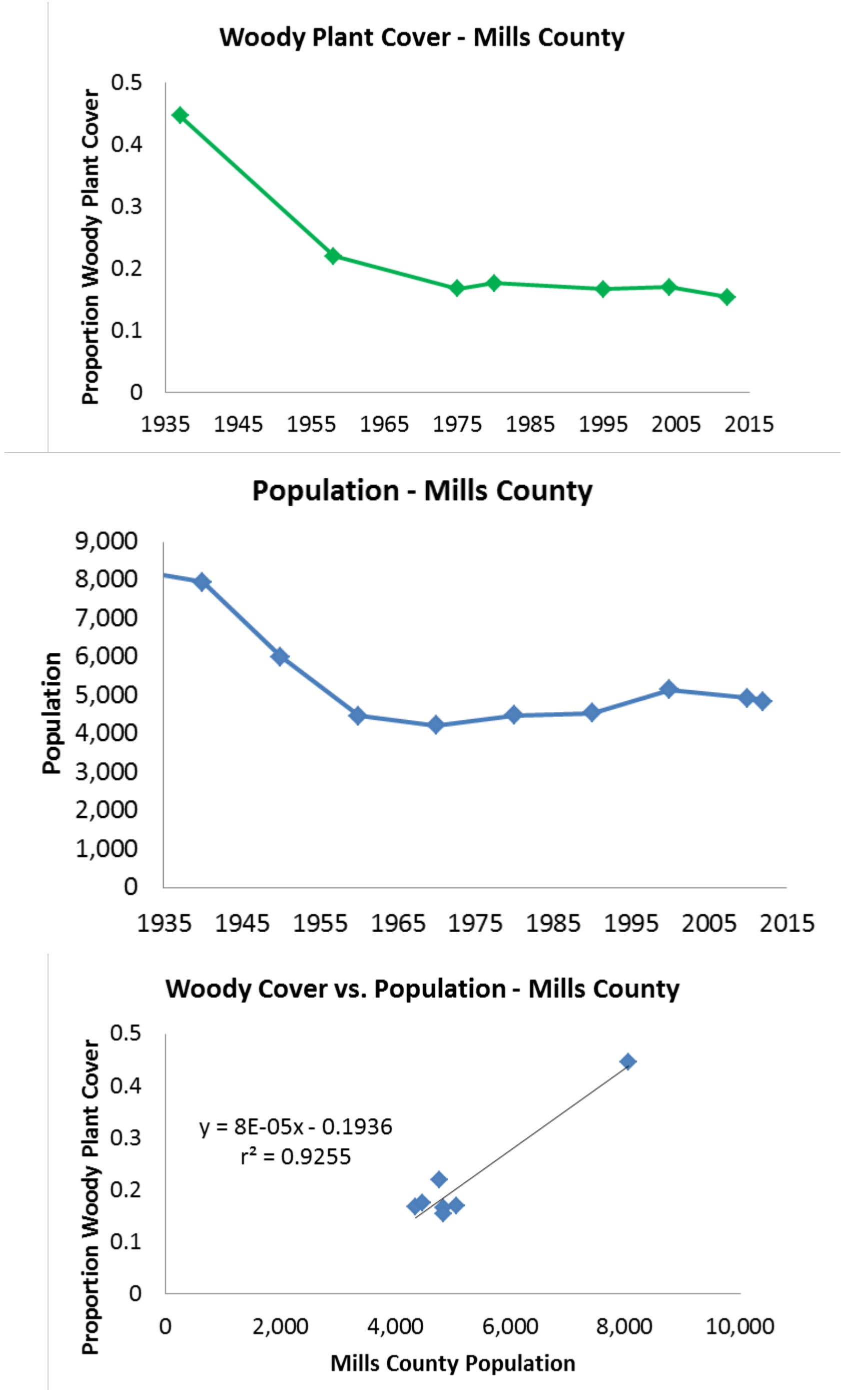


Figure 3.4. Trends in woody plant cover (top) and population (middle) for watersheds in rural Mills County follow the same pattern and are closely correlated (bottom).

Discussion

There is clearly a very close connection between human population trends and woody plant cover in central Texas. What makes this relationship even more surprising is the strength of correlation between these two trends in both rural and urban areas as well as during periods of both increase and decrease in human population. This association offer strong support to the idea that shrub distribution and woody plant encroachment in the Lampasas Cut Plain is not primarily a function of ecological constraints but rather one of human interaction.

Humans have had a complex history with woody vegetation depending on geographical and temporal context, ranging from complete deforestation to afforestation of regions where woody plants have not occurred during any previous time in the historical record. The increase in shrub coverage in the Lampasas Site 9 watershed is clear in historical imagery, and its relationship with increasing human populations perhaps most intuitive. Already experiencing effects of urban development in 1941, the watershed was home to a number of houses as well as a municipal cemetery, which covered a large portion of the area. Over time, the urban footprint expanded, with more roads and structures accompanying the rise in population of Lampasas. Fields were subdivided and developed, with smaller land parcels appearing to dominate the landscape. Trees were planted around homes for landscaping purposes, and routine brush management in many areas effectively ceased. This trend continued and gradually moved outward from the urban core, converting more and more land from agricultural production to residential use. However, the relationship between woody plant cover and population in rural watersheds at first glance is puzzling, particularly given the difference in trends between Lampasas County and Mills County. Given the highly dynamic nature of woody plant cover and the patchwork of encroachment and brush management shifting significantly over time, it is very difficult to determine with certainty woody plant trends before the photographic record here. Even from decade to decade, dense stands of shrubs have been wiped out, only to see neighboring cleared fields overrun by woody plants, and this happened many times over the period of out study, in some cases occurring on multiple occasions within a single plot. Since most old-growth shrub clusters have been removed over time, dating the woody plant expansion that had come to characterize the study area by the 1930s and 1940s using currently available resources is challenging. However, there are a number of sources that can help shed light on historical conditions and the timeframe of shrub establishment.

Analyses of soil organic carbon indicate that both live oak and Ashe juniper have been dominant residents in central Texas from long before European settlement (Jessup *et al.*, 2003). Interestingly, while oak mottes appear to be somewhat stable over time, coverage by juniper over time has been much more dynamic, with isotopic signatures of present shrub areas indistinguishable from grassland profiles at greater soil depths. While this may indeed point to recent woody plant encroachment, it may also indicate a complex patchwork of juniper cluster in constant flux due to disturbance. In the middle of the 20th century, Huss (1954) conducted a survey of Ashe juniper in the nearby Edwards Plateau, finding that the majority of shrubs standing at the time became established between 1854 and 1884. This indicates an age structure for the local community that is heavily skewed toward older plants, as well as an apparent period of episodic recruitment. Earlier still, Bray (1904) proposed in a report by the United States Bureau of Forestry that the Edwards Plateau and neighboring regions harbored such an extent of woodland resources that the region warranted classification as timberland. In many locations "cedar brakes" were so dense that they prohibited travel. Extensive stands of Ashe juniper supported growing cities through the supply of wood for many uses. Indeed, Bray mentions that fuel wood, fencing, telegraph and telephone poles, railroad ties, and construction materials already had been widely used for many years by the turn of the 20th century. At the same time, some stands were being cleared for the second time since initial European settlement. Ashe juniper was viewed as a valuable resource rather than an ever-present nuisance. Finally, several early historical accounts associated with colonial and military expeditions describe the ecological character of the area. Many descriptions from the first half of the 19th century tell of a broad landscape that was almost perfectly divided between grassland and woodland, with cedar brakes again so dense they made travel virtually impossible (Nelle, 1997b, Weniger, 1984). The Aguayo expedition of 1721 crossed Lampasas County and recorded a mixed landscape of prairies with numerous dense shrub stands in much of central Texas at a time when many native peoples inhabited the area (Buckley, 1911). It is clear that woody plants have been a major component of the central Texas landscape for centuries, and regrowth of Ashe juniper has occurred multiple times in single fields over the last three centuries despite the perception that recent increases are a novel condition.

To really understand the relationship between juniper and other woody species with human populations through time, it is critical to start with those first in time; the complex and dynamic communities of native people groups in central Texas. At the time of initial European settlement, the Lampasas Cut Plain was home to Comanche peoples, who were preceded in this landscape by the Tonkawa, who in turn followed after other indigenous groups in a dynamic anthropological history that predates the written record (Hasskarl, 1962). Many of these initial inhabitants employed controlled burns to improve hunting, crop management, many other uses (Lewis & Bean, 1973). Though specific records of native use of fire in Texas are rare, increasing evidence indicates it is certain that native peoples utilized fires to maintain and manipulate their surroundings (Smeins *et al.*, 1997). In many cases, this contributed to the maintenance of grasslands at the expense of woodlands. However, once European settlers drove prior inhabitants from the area, landscape dynamics shifted dramatically, leading to an increase in woody plant cover in many areas due to the removal of fire from ecosystems (Smeins *et al.*, 2005, Weniger, 1984). In places prescribed fire has now been absent functionally for more than 150 years, and the regional plant community has responded. In other portions of southern Great Plains, the conversion of prairie to woodland has been linked to long-term fire suppression efforts driving apparent shifts in species distribution at larger scales (Fuhlendorf & Smeins, 1997, Huss, 1954, Twidwell *et al.*, 2013a).

In central Texas, a popular public perception is that woody plants are relative newcomers on the landscape. This is particularly true of the opinion of Ashe juniper, which is viewed as an especially undesirable recent invader. In reality, this species has always been a dominant presence on the landscape and probably represents the climax community for many parts of the region (Nelle, 1997b). Furthermore, even many historical accounts of sweeping grasslands in the area likely are due to the lingering effects of long-term vegetation management dominated fire use by native peoples during prehistoric times (Huss, 1954). Even when fire was a more active force in the landscape, trees and shrubs were widespread and quite common across much of Texas. In fact, Weniger (1984) argues the idea that most of central Texas once was treeless grassland is "one of the most extraordinary of modern myths about Texas as it was."

Yet as the first European settlers arrived in the area, they represented a critical threshold in land management priorities and practices that encouraged a high degree of ecosystem change. After initial periods of heavy timber harvest, woody plant management largely ceased, led by the almost complete elimination of controlled burning. While mechanical brush management using manual methods was quite widespread (Hamilton & Hanselka, 2004), efforts could not keep pace with the expanding cover of shrubs and trees. As a result, woodlands began to reclaim much of the territory they had held at various points in the past. This contrast in landscape objectives was captured succinctly the observation by Pyne (1982) that "most of the impenetrable woods encountered by explorers were in bogs or swamps from which fire was excluded; naturally drained landscape was nearly everywhere burned. Conversely, almost wherever the European went, forests followed. The Great American Forest may be more a product of settlement than a victim of it." Suppression of fires associated with European settlement likely has led to the expansion of existing oak mottes but also to competitive advantage of other woody species such as junipers (Abrams, 1992, Engle, 1997). The counties of the Lampasas Cut Plain his effect has spilled over to drier areas as well. Even in south Texas, woody plant species were somewhat abundant as early as the 1820s, though they were somewhat more diffuse than in the present day. The primary reason for the localized expansion of existing populations has been a change in fire regime allowing individual plants to attain larger stature and generate a greater level of canopy cover (Johnston, 1963). Most of these settlers in the 19th century engaged in cropland agriculture to a much larger scale and in a much larger manner than the preceding inhabitants, and it is this permanence that discouraged both controlled burns and wildfires and allowed a rebound of Ashe juniper, oak, and other woody species that were once present in greater abundance.

The counties of the Lampasas Cut Plain were already major regional players in wheat production by 1860, and population continued to grow rapidly over the next several decades as the crop mix expanded and shuffled among wheat, cotton, oats, sorghum, and other crops (Jordan-Bychkov *et al.*, 1984). As in other portions of the country, land clearing for agriculture and abandonment of existing cultivated fields often occurred concurrently and in close proximity (Hart, 1968). As part of an extension of a trend that began in the eastern United States a few decades prior, cropland area eventually peaked across much of Texas and Oklahoma between 1910 and 1930 before undergoing decline and abandonment (Coppedge *et al.*, 2001, Ramankutty & Foley, 1999b). Population peaked at approximately the same time, and cropland bandonment continued, paired with significant socioeconomic changes for the next three to four decades. Even at the scale of global analysis of cropland changes since 1700, the abandonment of cultivated lands in a small part of the southern Great Plains occupied by the Lampasas Cut Plain is visible over the period 1940-1960 (Ramankutty & Foley, 1999a). Nearly all rural counties in the southern Great Plains experienced a loss of population greater than 30% between 1930 and 1970, and every county in the eastern two thirds of Texas lost more than 30% of its cropland area, with many experiencing abandonment rates of more than 65% (Roberts, 1987). In fact, urban counties were the few that actually grew over this period. Both Lampasas County and Mills County lost a large portion of their inhabitants, many of whom were tenant farmers, as residents relocated to larger cities elsewhere over this period.

At the same time, with greatly reduced available labor and changing economic conditions, abandoned croplands increasingly were acquired by ranchers with large livestock operations, representing another critical threshold in land management priorities (Conner, 2013). Much of this conversion was driven by declining productivity of rangelands that forced the purchase of additional grazing lands just to maintain the same number of animals within a herd (Box, 1967). Very high stocking rates in the 1800s had led to severe degradation of many rangelands and a subsequent reduction of well over half the animals in the Lampasas Cut Plain by 1910. As livestock production once again expanded to fill the void of abandoned cropland, most of the increase was among sheep and goats (Wilcox et al., 2012). Agricultural lands that have been in cultivation for shorter periods face fewer hurdles in returning to a previous condition (Hobbs & Cramer, 2007). When one traces the time since initial settlement to abandonment of croplands in Lampasas and Mills Counties, the vast majority of fields likely were in active production for only a few decades. When rangelands replaced cultivated fields, the removal of consistent mechanical disturbance altered the potential for establishment and survival of woody plant species. However, as rangelands replaced croplands, woody plant cover across the area began to decline. Landowners conducted a great deal of brush management activities that resulted in a steady loss of shrub cover from all local rural watersheds. As an indication of the widespread efforts to control regrowth juniper, the Bracero Program under the Mexican Farm Labor Program Agreement of 1942 provided temporary visas for migrant workers from Mexico to provide labor on farms across the southern United States. A great deal of this labor was dedicated to the removal of juniper in central Texas, including in the watersheds examined here (Conner, 2013, Garriga *et al.*, 1997). This program existed from 1942-1964, coinciding with the period of woody plant decrease seen across much of the area.

By 1970, cattle had overtaken sheep and goats as the primary livestock in the region due to changes in market conditions (Conner, 2013, Wilcox *et al.*, 2012). While browsing habits of goats tended to reduce Ashe juniper, live oak, and other woody species, cattle do not (Huss, 1954). It is at this juncture that the recent histories of Lampasas County and Mills County diverged. Mills County lost approximately half of its 1940 population by 1960 and has remained stable at that much lower level since that time. The same is true for woody plant cover in the area, which has exhibited the same trend of decline followed by stability. Though cattle densities have remained stable and relatively high over the last four decades, brush management has been very apparent in remote sensing imagery over this time, resulting in a net result of minimal change. However, the timing of this shift in grazing livestock and associated forage preferences corresponds with the timing of dynamics of both woody plants and population in Lampasas County watersheds, where total shrub cover and census data alike reversed decades-long declines in approximately 1970 and began to increase across the area once again. The decline in rural population, particularly in Lampasas County, follows a nearly identical trend in rural population across the whole of Texas. In this scenario repeated in many regions of the state, a peak was reached in 1940, followed by steady decline until 1970, at which point populations began to rebound to reach levels higher than those of 1940 in the present day (Jordan-Bychkov *et al.*, 1984). Livestock grazing is often a strong driver of shrub encroachment of semiarid regions (Grover & Musick, 1990). While the predominant livestock may have a role in explaining vegetation changes, other socioeconomics factors appear to be dominant in the Lampasas Cut Plain, as the recent expansion of woody plant cover is among species that were already historically abundant at certain locations in the landscape and tracks very closely with population changes in the area.

Prior to the mid-1970s, all brush on rangelands generally was considered undesirable and deserving removal (Welch, 1991). However, changes in landowner identity and objectives soon changed that perception. The first portion of Interstate Highway 35 was completed in 1962 in Austin (Beaumont *et al.*, 2006), and the 1973 oil embargo greatly strengthened the energy economy of the state (Drummond *et al.*, 2012).

These events led to an increase in access to the rural lands of central Texas as well as economic means for those with little agricultural background to purchase rural lands for a variety of reasons. As a result, the land market of this area and the Edwards Plateau to the west began to boom around this time (Snow, 2000). By the 1960s, potential revenues from leasing of property for deer hunting was seen as a huge untapped market that offered much greater return on investment than the production of livestock (Ramsey, 1965). At the same time, interest in outdoor activities increased rapidly and personal income devoted to recreation reached all-time highs, only to continue to increase.

This represents yet another critical threshold in land management priorities of the Lampasas Cut Plain, though this change appears to be much more dependent upon specific geography. The population and woody plant rebound that have occurred in Lampasas County have benefitted from the intersection of three U.S. Highways and proximity to Interstate Highway 35 as well as large urban centers, including Austin. Rural areas of Lampasas County are much more accessible to recreational traffic than are those in Mills County farther to the north. As a result, there may important differences in land use and economic models even over this short distance, and these may in turn be linked to increasing distance from nearby urban centers (Conner, 2013). A description of characteristics for Lampasas County indicates that commercial leasing for hunting purposes is widespread in the area, though no mention is made of similar activities in Mills County (Allison, 1991, Clower, 1980). Certainly recreational hunting is widespread in both areas, but the differences in proximity to major cities and highway infrastructure may be an important factor in land valuation and use. Changing land ownership characteristics have resulted in the loss of mid-sized parcels that have been subdivided into many smaller properties. In many cases, this has been linked to the previously mentioned urbanization and transportation infrastructure. The fragmentation and acquisition of properties by individuals with sometimes limited prior experience in land management has led to a great diversity in landowner objectives and activities, even over small areas. This has important but complex implications for ecosystem structure and function (Snow, 2000, Wilkins, 2000). Wildlife production for supplemental income as well as recreational purposes has seen tremendous growth in almost all Texas rangelands over the same period. Landowners prioritize management activities all along a spectrum from strict emphasis on livestock production to pure focus on wildlife enhancement, and these goals are sometimes at odds with one another (Rollins, 2000, Silvy, 2000, Sorice *et al.*, 2012). Recent property trends and the growing prevalence of recreational landowners indicate a premium placed on the presence of an intermediate amount of woody plant cover. Much of this interest is based on food and cover benefits to wildlife and even for purely aesthetic reasons (Bierschwale, 1997, Nelle, 1997a, Rollins & Cearley, 2004). In addition to withholding brush control efforts to improve recreational land values, the very wildlife that landowners appreciate likely play a role in woody plant dynamics. Numerous mammals and at least 15 bird species have been documented to consume Ashe juniper berries, with important implications for seed distribution (Chavez-Ramirez & Slack, 1994, Smeins & Fuhlendorf, 1997). At the same time, wildlife populations of some key species have rebounded over historical levels that were marked by wholesale hunting of low-value species for both food and sport by early settlers, providing further vectors for dispersal (Fleharty, 1995).

In addition to influencing overall woody plant cover and abundance over time, there is evidence that changes in landowner priorities and management have led to long-term shifts in the species composition of existing Lampasas Cut Plain woodlands. Especially in Mills County, live oak historically was a much more dominant presence among the woody plant community (Conner, 2013). Live oak is often seen as a preferred cover type due to is growth form, wildlife value, and perceived lesser effects on water resources (Rollins & Cearley, 2004). However, many oaks (*Quercus* spp.) are considered early- to mid-successional species that benefit from disturbance, namely fire (Abrams, 1992). In oak-dominated systems, wildlife abundance and diversity are related to the prevalence of fire in the ecosystem, with obvious implications for animal harvest (Engle, 1997). Indeed it is thought that many oak savannas developed from denser oak woodlands that underwent frequent burning, including that by native peoples. With increasing passage of time since the application of fire, existing oaks face growing pressure from later successional stages. While much oak reproduction is vegetative, animals can spur some recruitment by distributing acorns. But at certain densities, wildlife can hinder reproductive success. Present rates of deer harvest are probably much lower than that during initial periods of settlement, resulting in very high animal numbers in some areas, and intense grazing pressure may have serious negative consequences for oak recruitment, facilitating a shift to juniper dominance (Russell &

Fowler, 1999). Thus, wildlife numbers, related to changing human land uses and values, both enhance juniper encroachment and limit oak recruitment.

Thus, rural portions of Lampasas County and Mills County have been the recipients of differing socioeconomic factors that have led to very different land use and land cover characteristics over the last 50 years. The same is true of the Lampasas Site 9 watershed and Lampasas itself. While rural populations may contract, the core areas of towns and small cities generally do not, and peripheral growth around urban centers is responsible for a slow rate of growth at regional scales (Drummond *et al.*, 2012). As such, the city never experienced the population depression of the rural areas just a couple miles away. Furthermore, the transition from cropland and livestock production to urban use occurred very early in the settlement of the area, accompanied by much more permanent structural changes in land management. This eliminated the potential for subsequent major changes in the management of large tracts, which had already been subdivided and developed. Population and woody plant cover generally have continued to increase in this area, though at somewhat low rates in the last few decades. At some undetermined point, both woody plant cover and population likely will reach saturation points beyond which the correlation between shrubs and population begins to break down. Woody plants will necessarily be included from much of the urban area, representing an upper limit on its expansion. Additionally, city population is unlikely to undergo rapid increase without major changes in development patterns and peri-urban land use, which likely would eliminate some portion of existing woody plant cover. Still, among the oldest homes in the city, large, longstanding trees provide a great deal of cover especially at small scales.

As land uses and associated populations change, especially in rural areas, more than just agricultural land or wildlife habitat is lost. Population loss associated with cropland abandonment is accompanied by a loss of traditional farming knowledge and its associated familiarity with historical land management practices for a given area. This compounds the difficulty of returning previously abandoned lands to cultivation and serves as an obstacle to developing new farming professionals (Hobbs & Cramer, 2007). This trend is repeated in many contexts and different sociological settings. Valuable information is lost through outmigration, and over time as populations rebound, an influx of new inhabitants overwhelms existing land use objectives and knowledge. As a result, the newer population has a somewhat lower familiarity with the landscape and its history. Furthermore, perceptions and expectations of the land may be weighted toward much more recent expectations rather than an earlier baseline condition. Consequently, a phenomenon of shifting baselines occurs such that recent states are considered normal conditions and any departure from that is unprecedented. In the case of the Lampasas Cut Plain, this has played a role in the collective perception that woody plants are new members of the landscape, when in fact they were widespread decades ago and even more abundant near the start of the 20th century. Only by much effort and through changes in land use priorities were juniper and other species reduced to the point at which changes could be seen as long-term increase. Similarly, the total area covered by dense stands of woody plants (> 30% cover) has actually declined on a statewide level since the 1960s, suggesting large-scale thinning or removal of densest woody plant communities (Welch, 1991). Nevertheless, the dominant perception is one of steady increase and thicketization. Where increases have occurred, they may be a combination of repopulation of previously cleared sites and some establishment in new areas.

Increases in woody plant cover are known from a variety of settings throughout Texas and across the globe. However, land cover assessments covering such a large area over such a long period of time are rare. We have identified key trends in woody plant dynamics in different landscape settings. Significant land use changes and even passive influences can be responsible for the development of novel ecosystems with unpredictable structural and functional dynamics (Hobbs *et al.*, 2006). However, this portion of the southern Great Plains may not accurately fit this description. Overall, large changes in land cover have occurred across the Lampasas Cut Plain and all of central Texas, and these are tied to a complex assortment of factors including human populations and land management priorities. However, given the historical record and evidence of prehistoric conditions, it is probable that the current phenomenon of woody plant encroachment is not novel but has occurred on this landscape multiple times in the past, likely even during transitions between different native populations.

The ecological implications of cropland abandonment may be considered along two major perspectives: that of the potential for restoration to some prior or natural ecosystem or that of the potential loss of certain ecosystem components that depended on agriculture and related activities for continued existence (Hobbs & Cramer, 2007). This appears to be a critical consideration for the Lampasas Cut Plain and hinges on the interpretation of “natural” condition. In fact, both of these views may be appropriate for this landscape. Surely many of the recent wheat and sorghum fields have been abandoned. But if one considers the prairies and meadows encountered by the first European settlers as abandoned and released from the fire regime of prior inhabitants, the picture becomes very different indeed. This has profound implications for how we view paired social ecological systems and the long-term consequences different landscape trajectories.

While reversing the recent local spread of woody plants in the Lampasas Cut Plain and elsewhere in the southern Great Plains through manipulation of environmental conditions may be very difficult indeed, it appears a simple axe, grubbing hoe, or torch may be enough to make a lasting mark on the landscape, as has been the case for centuries and millennia past. Or maybe changing priorities may mean embracing the increase in juniper altogether.

CHAPTER IV

CONSTRUCTED SMALL PONDS PROLIFERATE WITH RURAL LAND USE CHANGE IN CENTRAL TEXAS

Introduction

Constructed ponds are important landscape features in many parts of the world and are utilized for livestock water, irrigation, recreation, sediment retention, aquaculture, and water quality control. Particularly in regions where they are abundant, small impoundments play a disproportionately large role in hydrological, sedimentological, geochemical, and ecological processes in watersheds of various sizes (Smith *et al.*, 2002, Verstraeten & Poesen, 2000). This influence is especially pronounced in rural areas, with up to 6% of international farmlands covered by small low-tech impoundments. In many cases, constructed ponds are closely related to agricultural activities, whether the production of crops or livestock. Global analyses of impoundments typically focus on large dammed reservoirs, and while these indicate total impoundment coverage of over 260,000 km^2, inclusion of small constructed water bodies adds more than 77,000 km^2 in area (Downing *et al.*, 2006, Graf, 1999). This number continues to rise over time.

In most parts of this country, small ponds are located on private lands and were constructed through a combination of landowner initiative and cost-share programs in cooperation with state and federal agencies, including the USDA-Natural Resources

Conservation Service (NRCS). These have become prominent features across local landscapes, clearly visible from both aerial imagery and on-site observations.

While small ponds are much smaller than more rigorously studied larger lakes and reservoirs on an individual basis, together they make up the vast majority of surface water bodies around the globe. However, effectively identifying and documenting small ponds remains a challenge due to the sheer magnitude of their numbers as well as their changing characteristics over time and difficulties in obtaining direct access. Very few studies have investigated the density of small ponds in the United States, and these generally have been regional in scope. In many nations, there has been no survey of these landscape components whatsoever. Where they have been conducted, undercounting these features has been a common problem in many surface water analyses, and this especially true of manmade water bodies in agricultural areas, which may increase 1-3% annually in parts of the United States (Chin *et al.*, 2008, Downing *et al.*, 2006, Smith *et al.*, 2002). Some estimates suggest there may be nearly 9 million small manmade ponds in the conterminous United States, representing a large proportion of the total surface water area in the nation (Renwick *et al.*, 2005b). Indeed, more than 2.1 million ponds were been built between 1980 and the first years of the 21st century alone, and due to its large geographical area, Texas leads the nation in total number constructed (Tuttle, 2003).

According to Renwick *et al*. (2005b), the combination of small impoundments serving as sediment sinks and accelerated erosion related to agricultural activity are among the most important human modifications to the geomorphic and hydrologic character of the United States, yet the timing and magnitude of these effects remain poorly understood.

In this study, our objectives are to 1) quantify the distribution and abundance of constructed small ponds on agricultural lands in the Lampasas Cut Plain of central Texas, 2) describe maintenance efforts including excavation and removal of existing ponds, and 3) explore the relationship between pond installation and agricultural land use change. We will perform this analysis using aerial imagery representing the last 75 years to identify periods of highest rate of pond construction and improvement and cropland abandonment. We expect to reveal a steady increase in pond density across eight watersheds, with greatest installation rates during and immediately after periods of severe drought. We also anticipate identifying these same periods as having the highest rates of excavation, with existing ponds more accessible for maintenance as a result of low water levels. It is believed that cropland area in the area has seen considerable declines throughout the period, leading to land use changes that favor pond presence.

Materials and Methods

Location Description

We investigated two clusters of watersheds in Lampasas, Mills, and Burnet Counties in the Lampasas River basin of central Texas. For this study, we examined six different watersheds located in the Sulphur Creek drainage west of the City of Lampasas and two in central Mills County on the Bennett Creek drainage. Median area of Mills County watersheds is $31.30 km^2$ and $25.36 km^2$ for those in Lampasas County. These areas enclose intermittent streams that supply flow to federally-funded Flood Control Structure (FCS) reservoirs impounded between 1958 and 1974. With the exception of very small Lampasas Site 9 partially within city limits of Lampasas, watersheds are rural in character. The predominant land use is rangeland, with livestock production widespread. The region is considered part of the Lampasas Cut Plain, transitional between the Edwards Plateau to the south and west, Cross Timbers to the north, and the prairies of north Texas (Allison, 1991, Clower, 1980) . This landscape features mesas and low buttes alternating with broad flat valleys and is dominated by a mosaic of woody plants and herbaceous species. Mean annual precipitation ranges from 710 mm in Mills County watersheds to near 780 mm in parts of Lampasas County.

Data Sources

We acquired a mix of aerial and satellite imagery from the earliest period available and then approximately once a decade to the present. Images were obtained from the USDA-Aerial Photography Field Office (APFO), Texas Natural Resource Information System (TNRIS), National Agriculture Imagery Program (NAIP), and Tobin International, Ltd., a private firm. As coverage across the area was not uniform over time, a few instances required merging imagery from consecutive years. Imagery represented years 1937/1940/1941, 1958, 1974/1975, 1980/1982, 1995, 2004, 2008, 2010, and 2012 and represented a combination of black and white, color infrared, and natural color imagery. Where necessary, hard copies were digitally scanned, and all images were radiometrically corrected and georectified to 2004 NAIP imagery (<1m error). We then resampled all images to a common 1m resolution using nearest neighbor methodology to standardize analysis and maintain consistency in appearance of landscape features across resolutions from different images. A description of imagery characteristics is found in Table 4.1.

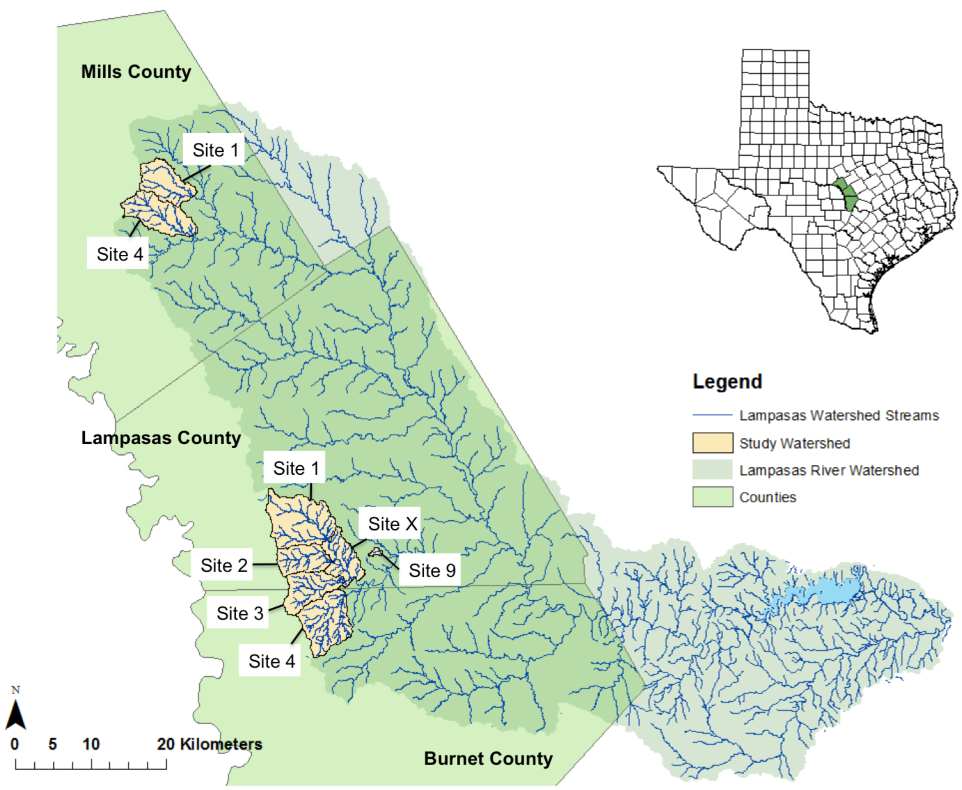


Figure 4.1. Map of study watershed location in Lampasas River Basin.

Table 4.1. Summary of remote sensing imagery characteristics.

Area	Image Year	Acquisition Date	Type	Spatial Scale	Spatial Resolution (m)
Lampasas Sites 1-4, 9, X	1940	December 1939-August 1941	BW	1:1667	0.45
	1958	December 1957-January 1958	BW	1:20000	0.24-0.51
	1974	February 1974	BW	1:40000-1:85000	1.01-1.02
	1982	December 1982	BW	1:4700	0.90
	1995	January 1995-February 1995	CIR	1:40000	1.01
	2004	September 2004-December 2004	CIR	1:40000	1.00
	2012	October 2012	NC	1:12000	1.00
Mills Sites 1,4	1937	December 1937-Novermber 1938	BW	1:1667	0.40
	1958	December 1957-April 1959	BW	1:20000	0.51
	1975	November 1975	BW	1:60000	1.02
	1980	November 1980	BW	1:2640	0.63
	1995	January 1995	CIR	1:40000	1.01
	2004	December 2004	CIR	1:40000	1.00
	2012	October 2012	NC	1:12000	1.00

Pond Identification

For each year, remote sensing imagery was assessed for the presence of constructed ponds within the FCS watersheds. While more recent imagery supports the identification of small ponds with a diverse array of spectral signatures due to differences in water clarity, aquatic plant abundance, and suspended sediments, early aerial images do not. Additionally, many ponds in this area are periodically dry and appear as bare ground for much of the year. For this reason, we used a visual approach to identify small ponds that may be misidentified by automated processes based on spectral characteristics alone. A stepwise process involving observation of tributaries followed by primary drainage and then uplands ensured exhaustive examination of each watershed area. For the purposes of this study, ponds were defined as any constructed landscape feature with the potential for retaining water and sediment, regardless of how much water they impound at any one time. Most ponds are constructed as such, but a raised roadbed that serves as a permanent barrier to flow would also be considered a pond in some circumstances.

We counted the number of ponds in each year and compared overall visual appearance of each feature to document maintenance activities. Significant changes in pond shape or size (controlled for varying water levels) and evidence of excavation were noted for each constructed pond as an evolving record of their function and condition. In cases where the features appeared no longer to retain water or sediment, they were not considered ponds and were noted as being removed.

Cropland Trends

To investigate the relationship between the construction of small ponds over time and the land use dynamics of the surrounding area, we visually indentified active croplands from time series remote sensing imagery. We classified agricultural fields on the presence of either: 1) evidence of harvest activities or 2) evidence of site preparation activities. In addition, we identified as croplands fields with uniform texture and spectral signature distinct from adjoining rangelands if they exhibited 1) zero evidence of encroachment by shrub saplings and 2) no sign of long-term grazing by livestock. As a result, hayfields were classified as active croplands, while grazed rangelands were not. We hand-digitized and delineated fields using ArcGIS 10.0 (ESRI, 2010) for each imagery year except 2008 and 2010. Following cropland identification, we calculated cropland area for watersheds in each Lampasas County and Mills County and performed correlation analysis between pond density and cropland area for these two areas.

Results

From the first year of remote sensing imagery, the overall number and density of constructed small ponds has increased dramatically in all watersheds. Pond number has increased threefold in Lampasas County watersheds and nearly fivefold in Mills County (Table 4.2). At the beginning of the study period, pond density in both areas was comparable (0.70 ponds/km^2 in Lampasas County, 0.89 ponds/km^2 in Mills County). However, by 2012 Mills County watersheds contained 4.25 ponds/km^2 compared with 2.07 ponds/km^2 in Lampasas County. The greatest increase in both areas occurred prior to 1958, though this increase was much more rapid in Mills County. Rates of increase in each county were similar and much more gradual after this time (Figure 4.2).

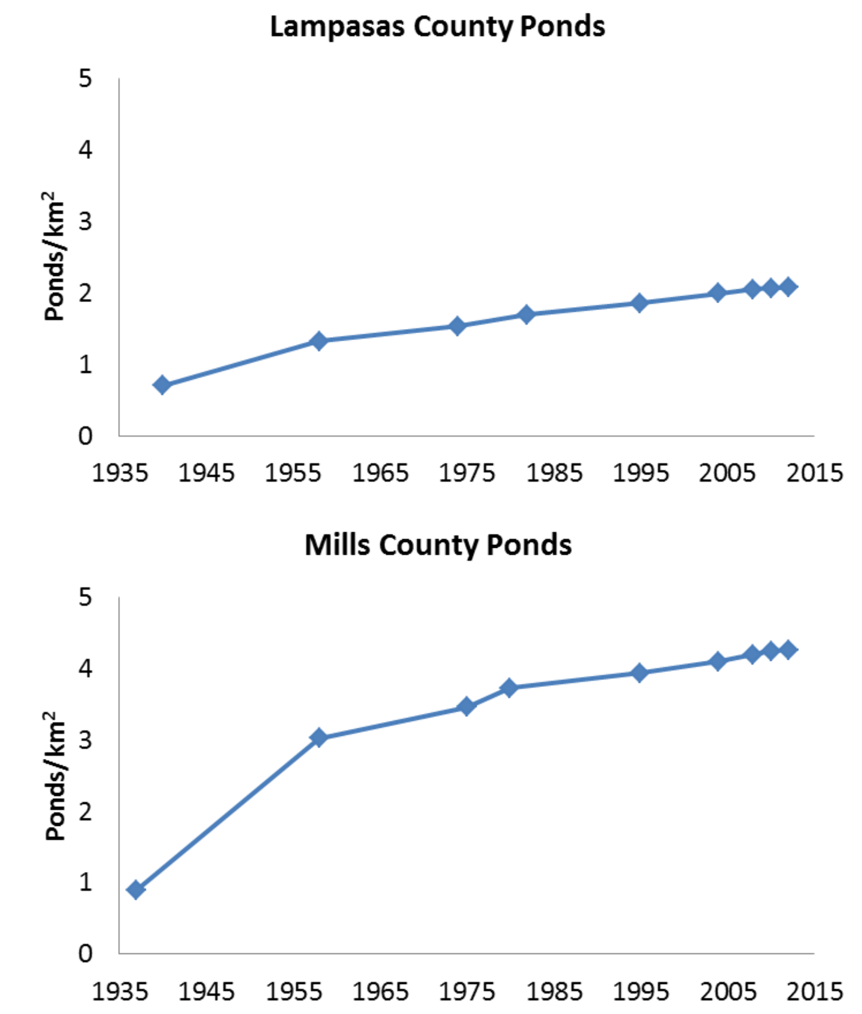


Figure 4.2. Increase in pond density over time. Recent rates of pond construction have been similar, though Mills County experienced a much higher rate of pond installation prior to 1958.

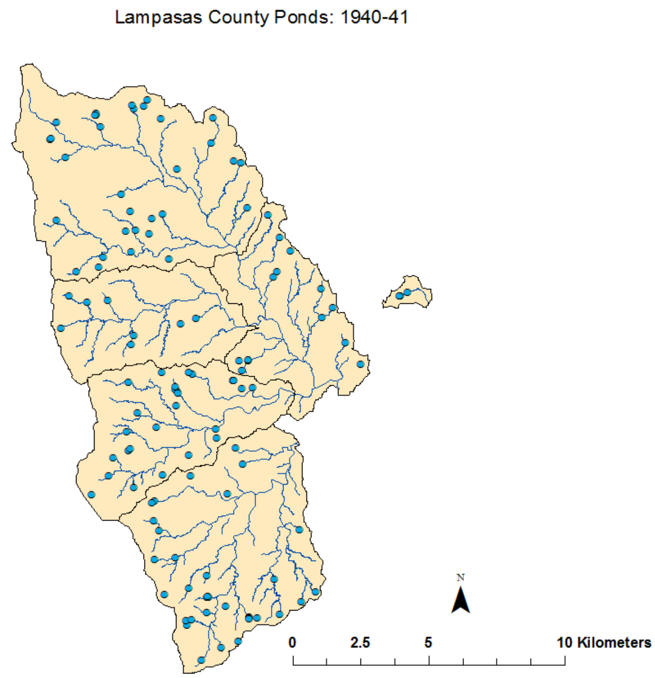
Table 4.2. Number of small ponds and density for each watershed over time**.**

		1937-41	1958	1974-75	1980-82	1995	2004	2008	2010	2012
Lampasas Site 1*	Count	32	63	88	103	111	127	130	130	130
	Density (#/km^2)	0.65	1.26	1.75	2.04	2.20	2.51	2.57	2.57	2.57
Lampasas Site 2	Count	8	20	20	22	24	26	26	27	27
	Density (#/km^2)	0.34	0.86	0.86	0.95	1.03	1.12	1.12	1.16	1.16
Lampasas Site 3*	Count	27	57	60	61	64	65	67	69	70
	Density (#/km^2)	0.98	2.07	2.18	2.19	2.33	2.36	2.44	2.51	2.55
Lampasas Site 4	Count	33	59	65	66	74	74	75	75	75
	Density (#/km^2)	0.80	1.43	1.58	1.60	1.80	1.80	1.82	1.82	1.82
Lampasas Site 9	Count	3	3	3	3	5	6	6	6	6
	Density (#/km^2)	2.42	2.42	2.42	2.42	4.03	4.84	4.84	4.84	4.84
Lampasas Lively	Count	14	18	20	27	31	34	38	38	38
	Density (#/km^2)	0.69	0.87	0.95	1.25	1.43	1.56	1.73	1.73	1.73
Lampasas Sites	Count	117	220	256	282	309	332	342	345	346
	Density (#/km^2)	0.70	1.32	1.53	1.69	1.85	1.99	2.05	2.06	2.07
Mills Site 1	Count	20	92	109	118	125	134	139	141	143
	Density (#/km^2)	0.58	2.66	3.15	3.41	3.61	3.87	4.01	4.07	4.13
Mills Site 4	Count	36	97	107	115	121	122	123	124	123
	Density (#/km^2)	1.29	3.47	3.82	4.11	4.32	4.36	4.40	4.43	4.40
Mills Sites	Count	56	189	216	233	246	256	262	265	266
	Density (#/km^2)	0.89	3.02	3.45	3.72	3.93	4.09	4.19	4.23	4.25
All Sites	Count	173	409	472	515	555	588	604	610	612
	Density (#/km^2)	0.75	1.78	2.05	2.24	2.42	2.56	2.63	2.65	2.66

*Due to image availability, very minor portions of coverage from 1940 and 1982 for Lampasas Sites 1 and 3 were omitted. In each case, the omitted area is less than 1% of the total watershed area. This did not affect pond detection.

From a landscape perspective, the proliferation of ponds is quite noticeable when viewed from the initial and final years of imagery. Additionally, the majority of ponds in Lampasas County watersheds by 2012 are located higher in the drainage area, on small tributary streams. This is in contrast with pond locations in Mills County, which generally are located on higher order streams and in lower portions of the watershed (Figures 4.3 and 4.4).

Both watershed areas experienced a great deal of pond excavation and enlargement through the years, with 98 of 359 ponds present at some point in time in Lampasas County (27.3%) and 113 of 276 in Mills County (40.9%) receiving significant maintenance at least once since the beginning of the study period. In Lampasas County, all of this activity occurred prior to 2004, with relatively high levels of pond maintenance preceding 1958, 1974, 1995, and 2008 (Figure 4.5). No ponds have been removed since 2004. In Mills County, pond excavation has been much less common in recent years, and pond abandonment was recorded only in 1995 and 2012 (Figure 4.6). Interestingly, many sites underwent excavation or enlargement on multiple occasions. In the Lampasas County watersheds, 17 ponds (4.7%) and 18 ponds in Mills County (6.5%) received maintenance at least twice, with three of those Mills County sites (1.1%) undergoing excavation three times. Since 1940-1941 a total of 18 ponds has been removed from Lampasas County watersheds, compared with 11 in Mills County



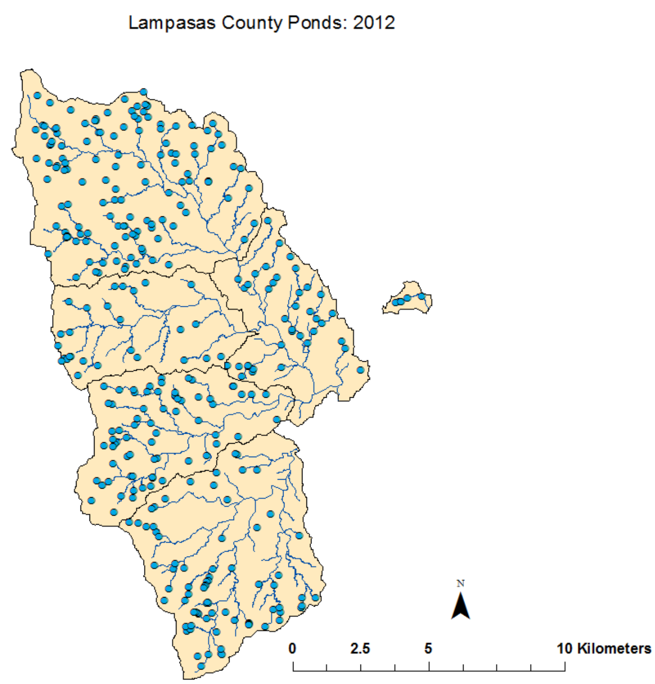
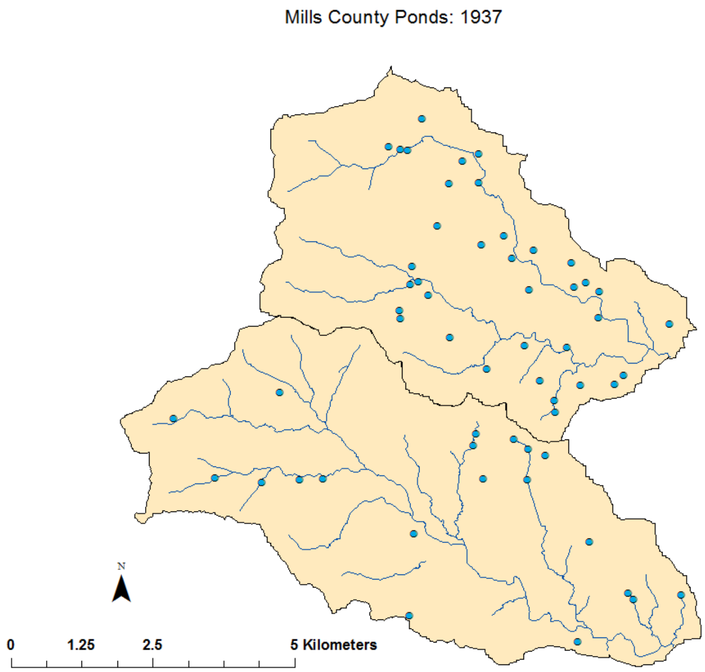


Figure 4.3. Small pond distribution for Lampasas County watersheds in 1940-41 (top) and 2012 (bottom). Pond abundance has tripled in this area.



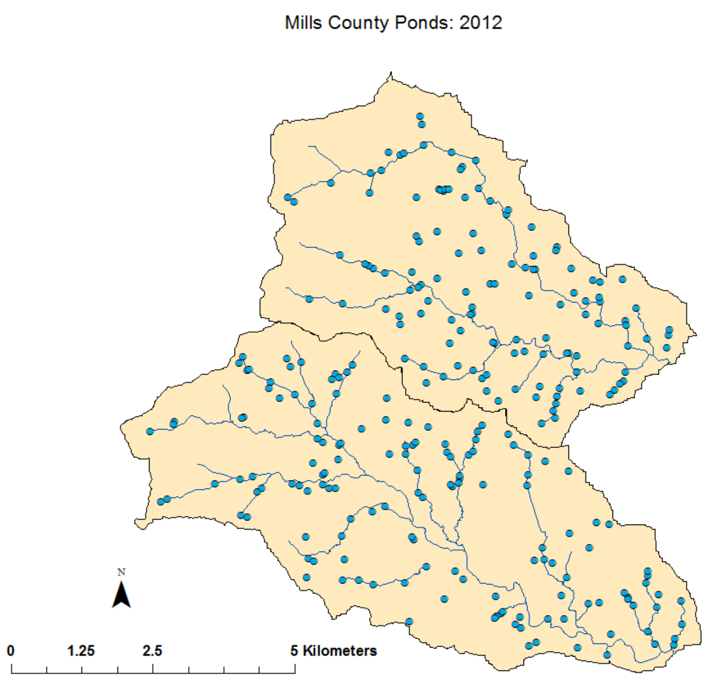
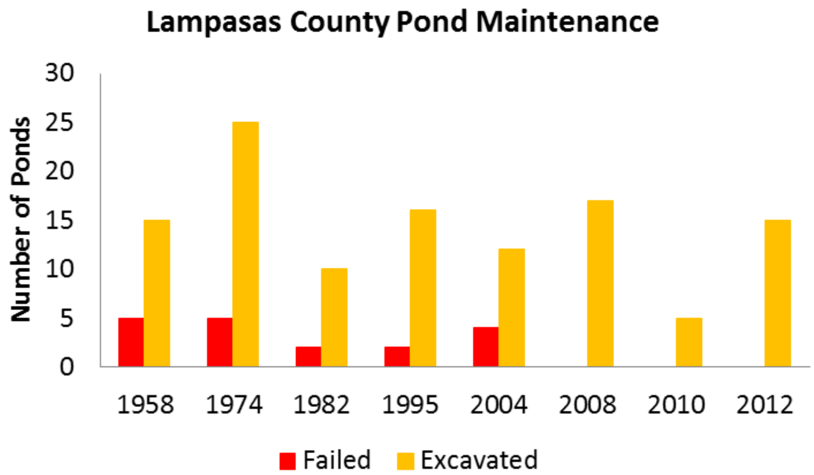


Figure 4.4. Small pond distribution for Mills County watersheds in 1937 (top) and 2012 (bottom). Pond abundance has increased 375% in this area.



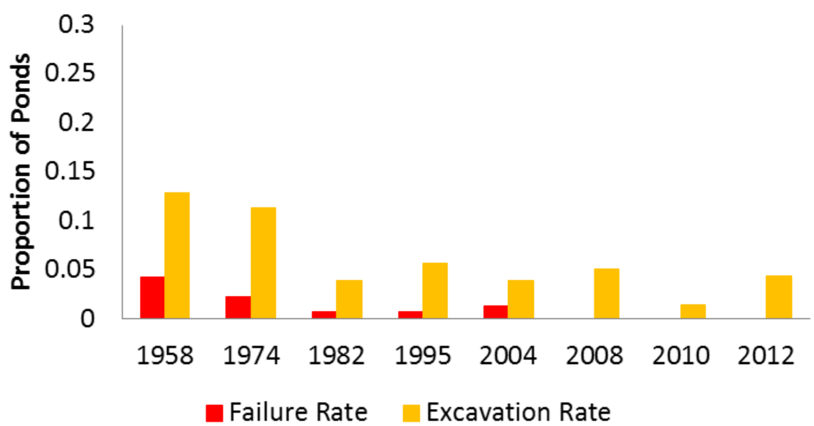


Figure 4.5. Number of small ponds undergoing maintenance activities (top) and rate of maintenance activities as a proportion of total ponds (bottom) in Lampasas County watersheds.

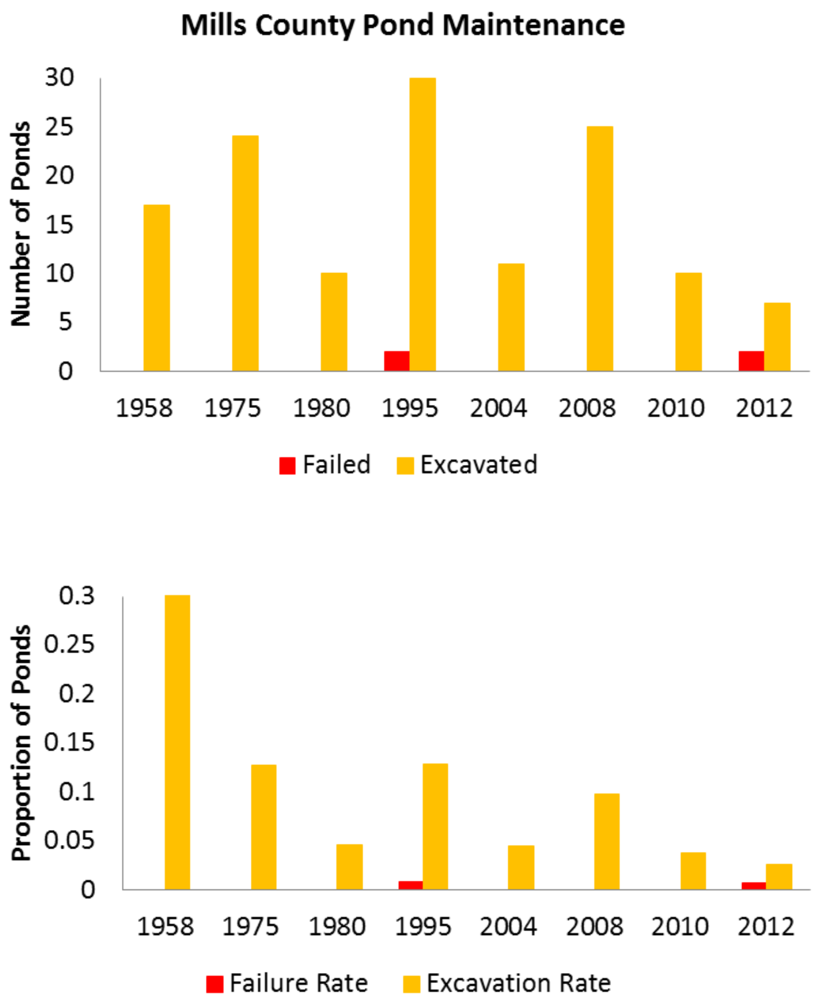


Figure 4.6. Number of small ponds undergoing maintenance activities (top) and rate of maintenance activities as a proportion of total ponds (bottom) in Mills County watersheds.

Table 4.3. Cropland by area and proportion of watershed for each watershed over time.

		1937-41	1958	1974-75	1980-82	1995	2004	2012
Lampasas Site 1*	Area (km^2)	6.72	5.76	5.33	4.02	4.49	2.76	2.65
	%	13.2	11.3	10.5	7.9	8.8	5.4	5.2
Lampasas Site 2	Area (km^2)	2.30	2.09	1.05	0.27	0.17	0.25	0.05
	%	9.9	9.0	4.5	1.1	< 1.0	1.1	< 1.0
Lampasas Site 3*	Area (km^2)	6.37	5.79	4.07	0.87	1.97	1.13	1.17
	%	23.4	21.1	14.8	3.2	7.2	4.1	4.2
Lampasas Site 4	Area (km^2)	4.63	2.85	1.85	1.44	0.99	0.46	0.37
	%	11.2	6.9	4.5	3.4	2.4	1.1	< 1.0
Lampasas Site 9	Area (km^2)	0.10	0.02	0.02	0.03	0.05	< 0.01	0.03
	%	7.9	1.9	1.6	2.3	4.3	< 1.0	2.3
Lampasas Site X	Area (km^2)	2.60	2.11	1.73	1.00	1.48	0.55	0.49
	%	11.3	9.1	7.5	4.3	6.4	2.4	2.1
Lampasas Sites	Area (km^2)	22.73	18.63	14.06	7.61	9.15	5.16	4.75
	%	13.6	11.1	8.4	4.6	5.5	3.1	2.8
Mills Site 1	Area (km^2)	9.41	7.63	3.51	2.43	2.96	2.13	1.79
	%	27.2	22.1	10.1	7.0	8.6	6.2	5.2
Mills Site 4	Area (km^2)	5.68	4.56	3.74	2.18	1.44	1.59	2.05
	%	20.3	16.3	13.4	7.8	5.2	5.7	7.3
Mills Sites	Area (km^2)	15.09	12.19	7.25	4.61	4.41	3.72	3.84
	%	24.1	19.5	11.6	7.4	7.0	5.9	6.1
All Sites	Area (km^2)	37.82	30.81	21.31	12.23	13.55	8.88	8.59
	%	16.5	13.4	9.3	5.3	5.9	3.9	3.7

*Due to image availability, very minor portions of coverage from 1940 and 1982 for Lampasas Sites 1 and 3 were omitted. In each case, the omitted area is less than 1% of the total watershed area. For all other years examined, no cropland was located in these areas.

Cropland Trends

The trend in small pond installation closely parallels widespread abandonment of cropland in the region. Since the late 1930s and early 1940s, most of these areas have ceased cultivation and undergone conversion to rangeland use. Lampasas County watersheds lost 17.98 km^2 of actively cropped fields since 1940, dropping from 13.6% to 2.8% as a percentage of watershed area (Table 4.3). This represents a decline of 79.1% in total area. In Mills County, 11.25km^2 of cropland since 1937, a decrease of 74.5%. Cropped area represented 24.1% of total watershed area at the beginning of the study period. By 2012, this had dropped to 6.1%. Lampasas County watersheds lost nearly their entire cropland area over this time, and nearly all remaining Mills County croplands were located in lowlands adjacent to streams by the end of the period (Figures 4.7 and 4.8). In both Lampasas County and Mills County, cropland abandonment was more rapid prior to the 1980s and accelerated in the few years immediately preceding 1980/1982 (Figure 4.9).

When compared with one another, a close relationship between cropland abandonment and the proliferation of small ponds is apparent. Pond density displays a strong negative correlation with cropland area in both Lampasas and Mills Counties (r^2 = 0.92 and 0.85, respectively, Figure 4.10). As croplands have decreased sharply over the last 80 years, pond density has done quite the opposite, with small ponds proliferating during the widespread conversion of land use in each county.

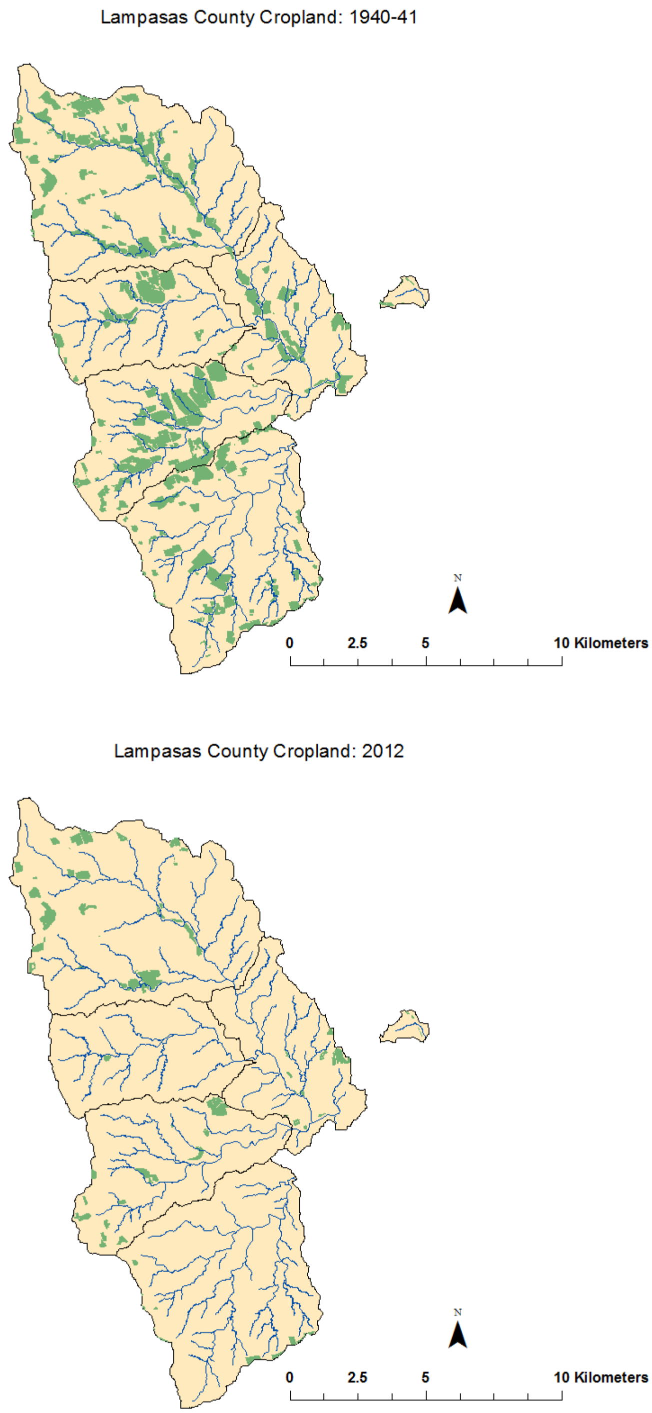


Figure 4.7. Cropland distribution in Lampasas County watersheds in 1940-41 (top) and 2012 (bottom). Cropland coverage has decreased nearly 80% in this area.

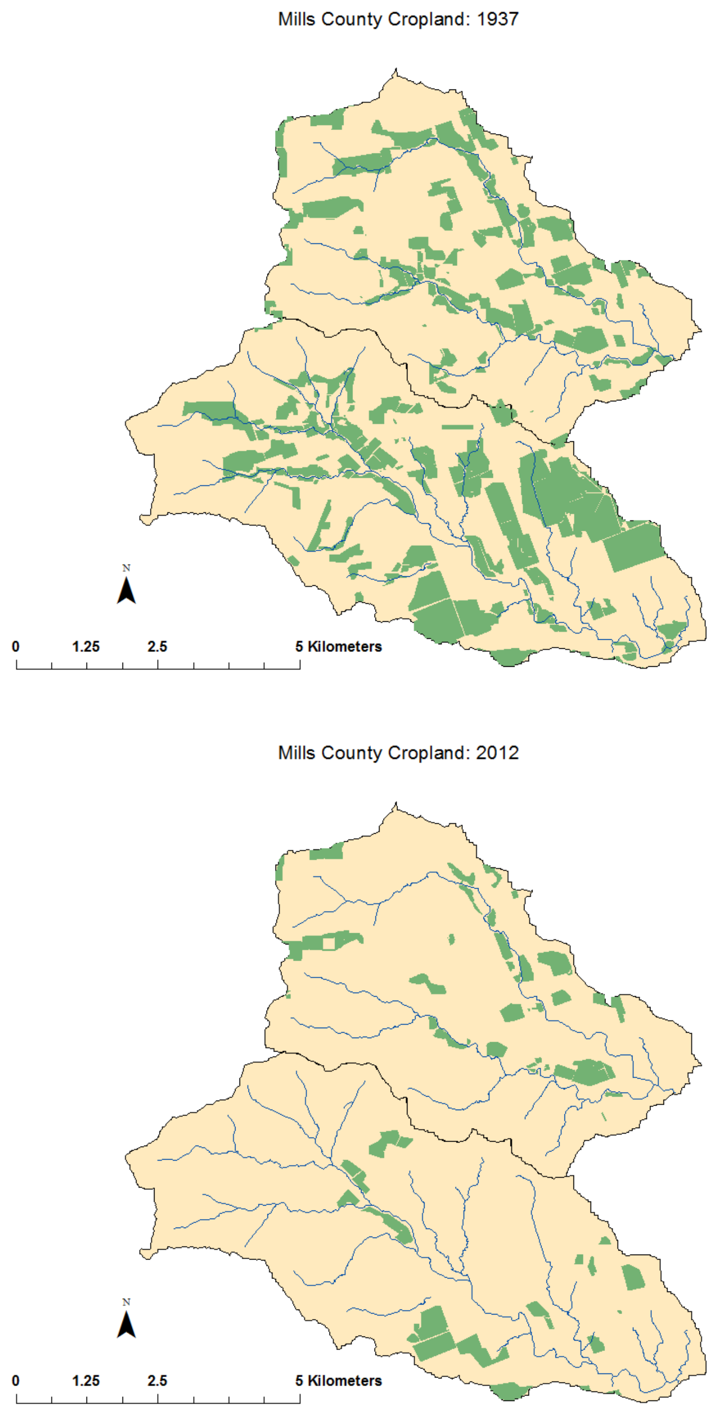


Figure 4.8. Cropland distribution in Mills County watersheds in 1937 (top) and 2012 (bottom)**.** Cropland coverage has decreased approximately 75% in this area.

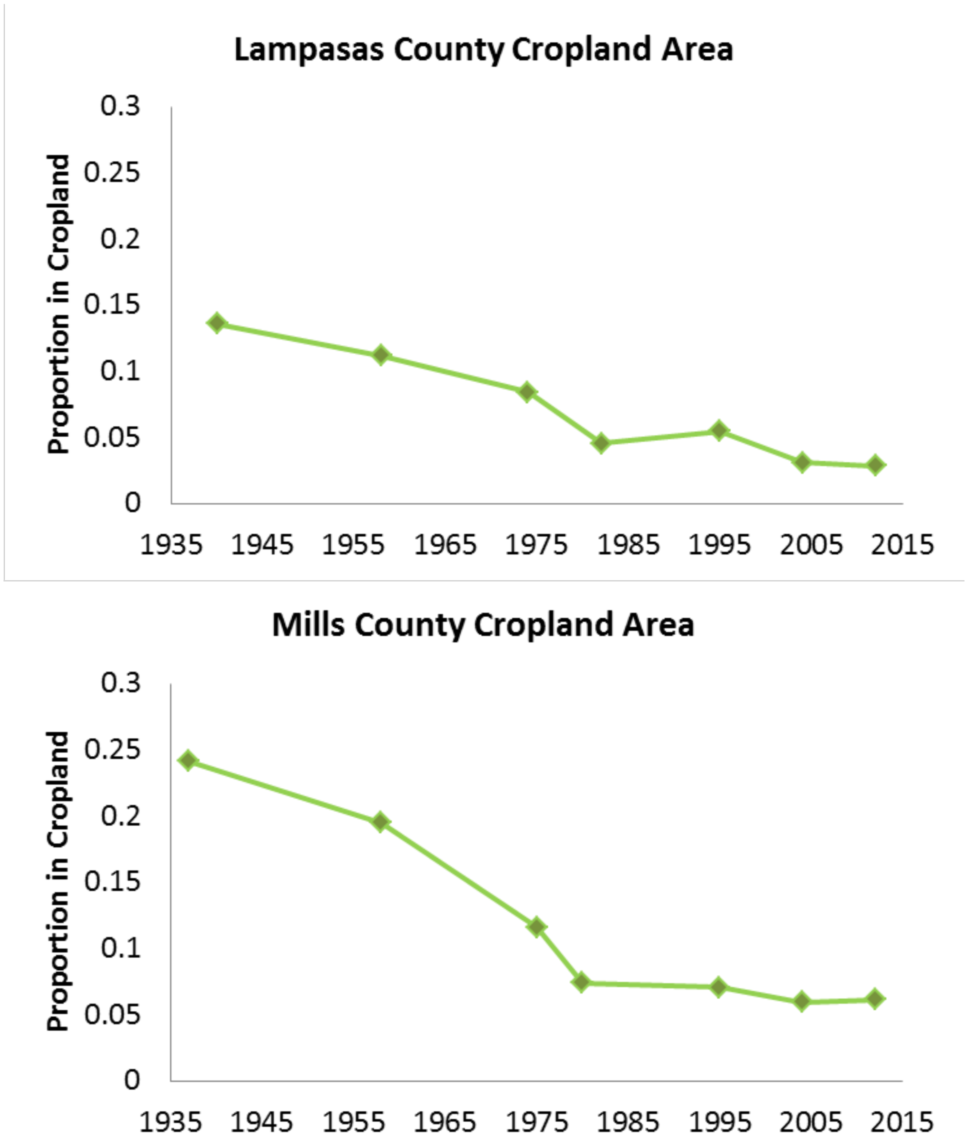
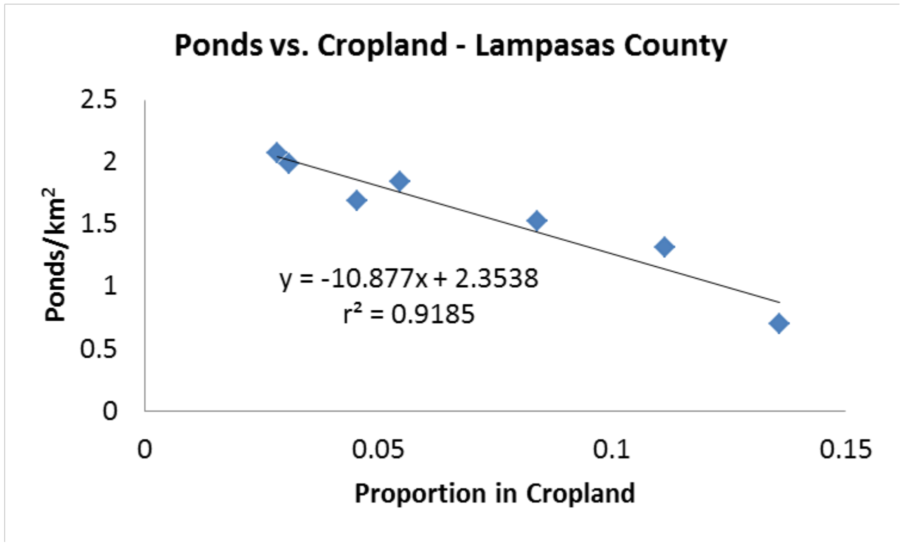


Figure 4.9. Cropland abandonment over time in Lampasas County and Mills County watersheds.



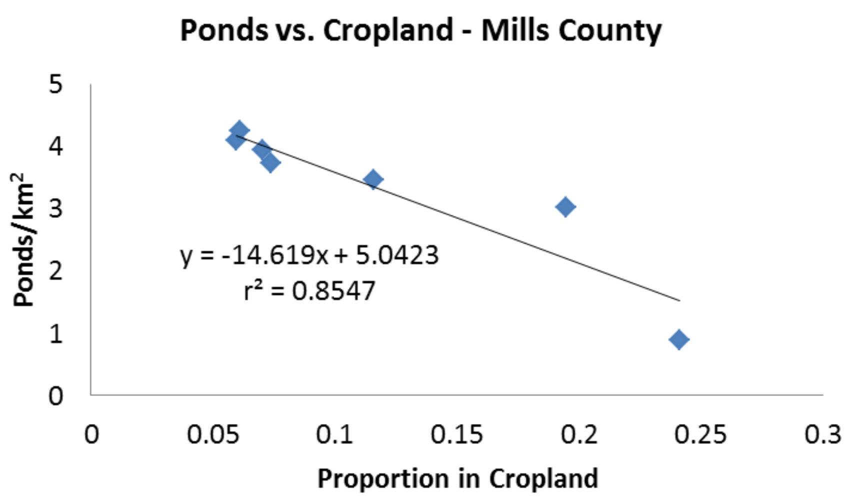


Figure 4.10. Correlation between cropland area and pond density in Lampasas County (top) and Mills County (bottom). These landscape feature are strongly correlated in both areas.

Discussion

The Lampasas Cut Plain has undergone a great deal of land use change over the past 80 years, and this is reflected in the extent an distribution of both cultivated fields and constructed small ponds. While trends of both landscape features have continued over the study period, rapid pond installation appears to have preceded the period of greatest cropland abandonment by approximately two decades, though both began early in the 20th century. Small ponds were constructed at greatest rates prior to 1958, while the loss of cropland area continued steadily until 1980. Each change is consistent with sweeping transitions that have occurred across much of the eastern United States over the past century.

At the national scale, land conversion for crop production began in the eastern United States and swept through the Great Plains to the west, accompanying expansion of European settlement and population growth. Already by 1860, Lampasas County had become important in regional wheat production (Jordan-Bychkov *et al.*, 1984). Over time, cotton, oats, and sorghum also fluctuated in importance. Mean annual precipitation generally is considered sufficient to sustain the dominant local crops (Allison, 1991, Clower, 1980). However, cropland acreage reached its maximum across the southern Great Plains in the period 1910-1930 (Ramankutty & Foley, 1999b). Since that time, large decreases in agricultural land area have occurred across much of the region and typically are accompanied by decreases in agricultural employment and population (Parton *et al.*, 2003).

Abandonment trends followed the same general geographical pattern as initial conversion for cultivation, migrating from east to west across North America. There appears to be a temporal lag in cropland abandonment related to the initial time the land was converted to agricultural use (Hobbs & Cramer, 2007, Ramankutty & Foley, 1999b), and much of the nation's cropland abandonment has occurred in areas previously covered by forests and woodlands. Between 1930 and 1970, every county in the eastern two thirds of Texas experienced losses of cropland area in excess of 30%, with many feeling the effects of abandonment rates over 65% (Roberts, 1987). Even at the coarsest of scales, global analyses of cropland changes highlight the loss of cropland in the counties of the Lampasas Cut Plain over the period 1940-1960 (Ramankutty & Foley, 1999a). This large reduction is verified by our observations, with more than 77% of the land in cultivation at the beginning of the study period had been lost by 2012. Continued expansion of cropland area has occurred in some small portions of the southern Great Plains. However, these localized gains are almost always due to irrigated agricultural production (Parton *et al.*, 2003). It stands to reason that the converse is also true; where dryland farming dominates, reductions in cropland are more likely. Irrigation is very uncommon in the Lampasas Cut Plain, and small temporary increases in cultivated area we observed likely are linked to short-term improvements in climatic conditions favoring production. Some of the recent stability in Great Plains cropland area has been linked to increased demand for hay production (Parton *et al.*, 2003). Indeed, the majority of remaining cultivated areas in Lampasas and Mills Counties currently are devoted to forage crops. Imbalance between precipitation and evaporative demand likely has contributed to a great deal of the cropland abandonment observed in the area, combined with long-term economic factors and degradation of croplands that were marginal from the outset.

Opposing the trend in cropland reduction, small ponds have proliferated across the study area in the last 75 years, though interpreting the effect of and mechanisms behind pond numbers is also complicated by changing cultural and economic factors, which influence local dynamics above and beyond purely environmental variables (Buddemeier, 2005). As croplands were converted to rangeland use in these watersheds, variable annual precipitation probably encouraged land managers to conserve and utilize what rainfall and runoff was received, and small impoundments have become more common over time. However, it is important to note that even in the late 1930s and 1940s, a relatively high number of ponds were already present. In fact, early descriptions of central Texas recognized the periodic need for additional water supplies, and many livestock producers set out to construct small impoundments for supplemental sources from very early periods (Bentley, 1898). In Alabama, counties with higher populations have slightly higher densities of small impoundments (Chaney *et al.*, 2012), and pond density has increased with population growth in parts of Pennsylvania (Fairchild *et al.*, 2012), though ponds tended to decrease in areas of dense urban development. The small Lampasas Site 9 watershed partially within city limits indeed harbors a higher density of ponds than other portions of the Lampasas Cut Plain. However, a rural nature and much lower populations characterize most of the region. By the date of initial imagery used in this study, pond densities already compared favorably with some of the higher densities across the nation observed in the present day (Smith *et al.*, 2002).

Though very few historical analyses of pond presence have been conducted, results from (Buddemeier, 2005) show pond densities in parts of eastern Kansas during the early 1940s were more than 20 times lower than those in our Lampasas Cut Plain study site. Despite those much lower initial densities, that region has gained a substantial number of ponds to become one of the highest concentrations in the nation, though still lower than those in central Texas. This suggests an important chronological component of the pond landscape that likely is very site-specific and dependent on local history. This is likely extremely significant when comparing trends between regions or even continents. Addressing primarily small public reservoirs, Chin et al. (2008) identified the 1950s as a period of high activity in the construction of small dams throughout the nation. In a different ecological and land use context, farm ponds increased rapidly in Pennsylvania over the same period and following years due to greater federal investment in conservation programs (Fairchild *et al.*, 2012). This appears to carry over into similar efforts to improve agricultural water supply in rural watersheds, as in Lampasas and Mills Counties. The period of greatest increase in pond density in our study area occurred in the first half of the 20th century. Our observations place the Lampasas Cut Plain among the highest densities of small ponds in the United States, as our results from Mills County surpassed even the maximum densities (3 ponds/km^2) recorded by Smith *et al*. (2002). The same analysis did find Texas to house the highest concentration of small ponds in the country, though this has much to do with the comparatively larger size of the state. Even when controlling for state area, a preliminary analysis by Downing *et al*. (2006) revealed that farm pond area as a percentage of all agricultural land area is somewhat higher in Texas as a whole than would be expected based on global relationships between small ponds and precipitation. Most small impoundments in the United States are located in the eastern half of the country where precipitation is more abundant, but within this area, the greatest density of ponds is in portions of Texas, eastern Oklahoma, eastern Kansas, and Missouri (Smith *et al.*, 2002). Thus, due to a combination of topography, land use, and climate, there appears to be a geographical "sweet spot" of pond installation where precipitation is moderately abundant but prone to large swings from year to year and where land activities lend to numerous, small water sources. In portions of the lower Great Plains frequently stricken by drought and home to an abundance of rangelands, additional surface storage in the form of these impoundments has become an appealing, even critical proposition.

The exceptionally high abundance of ponds in Lampasas County and Mills County suggests either a significant underestimation of small ponds by remote sensing techniques or an exceptional density of constructed ponds in this portion of central Texas. We suspect it is a combination of both. For instance, pond inventories employing automated classification have been shown to be very susceptible to climatic variation, with pond numbers declining in dry periods and fluctuating between years (Buddemeier, 2005). This represents a potential weakness in this approach; as ponds may indeed be present structurally and continue to play a role in local landscape processes yet hold no water. Indeed, many ponds in our analysis periodically went dry during long-lasting or severe droughts and would otherwise have gone undocumented by automated methods. We did not estimate water volume or surface area of water in these ponds due to extreme fluctuations between years and complete drying in drought periods. In reality, this dramatic variation in appearance likely represents a major factor in chronic underestimation of pond density, as remote sensing procedures would be incapable of automatically identifying a dried mud surface or a vegetated depression as temporarily dry ponds. An automated survey of small impoundments in Alabama yielded an accuracy rate of only 80%, meaning many ponds were not identified (Chaney *et al.*, 2012). Additionally, spatial resolution was found by Buddemeier (2005) to be a major factor in pond classification, with finer resolution enabling detection of many more sites than coarser imagery. Our resampled 1 m data supported a very high degree of detail that is not possible with 30 m Landsat ETM+ or other imagery. In addition, the number and location of small impoundments changes through time. On top of these obstacles, nearly all ponds are located on private property, making access for ground verification a challenge. Because of this, many sites are missing from even digital databases of hydrological features. Due to the exhaustive nature of our high-resolution manual approach, we are confident we identified all potential small ponds and classified them accordingly. However, this strategy was time-intensive, making comprehensive large-scale pond identification using high-resolution data sources more difficult.

There are likely some scaling effects with pond density at increasing scales. Small impoundments in this area are high in their watershed, near the divide of two major river basins where elevations and slopes are somewhat higher than in downstream areas. These characteristics make such areas prime candidates for pond construction, and it is likely that farm pond density decreases with increasing watershed size as lower portions of the basin with more gradual slopes are considered. Toward the end of the study period, there does appear to be a slight leveling of pond numbers, and this may be a function of saturation of ideal locations for small dams or of insufficient flow or catchment area downstream from existing structures to make construction of additional ponds worthwhile. Regardless, it is likely that new ponds will plateau at some point in the future, in which case excavation or enlargement of existing ponds might prove a more viable option.

We did not directly measure sedimentation of small ponds in these watersheds. However, visual assessment allows interpretation of excavation, enlargement, and abandonment events rather than mere fluctuating water levels, shedding additional light on sediment dynamics. One of the most readily apparent observations confirmed by our study is that ponds are far from static features. Over time they can be filled by human action or by natural processed of sedimentation, they are dredged to prolong lifespan and utility, they are enlarged to boost storage capacity, and they are occasionally abandoned or removed due to dam failure or intentional breaching to increase downstream flow. Buddemeier (2005) found 93% of ponds present in southeastern Ohio in the 1950s were absent by the 2000. Our rates of abandonment were much lower. This may be a function of lower sedimentation rates. However, the large proportion of sites receiving some excavation over time suggests a more active commitment to maintenance efforts in a region where precipitation is much more limiting and vital to agricultural production.

Our observations indicate very high rates of pond maintenance, which in many cases served to remove deposited sediments and restore previous storage capacity.

The increasing density of these constructed ponds presents a suite of important ramifications for watershed processes, as they slow runoff velocity and cause localized sedimentation (Verstraeten & Poesen, 2000). The number of small ponds has been shown to be a significant determinant of watershed sediment yield, potentially trapping large proportions of sediment that would otherwise be transported downstream to water supply reservoirs (Foster, 2011). Over time, trap efficiency of small ponds decreases as a result of storage volume loss and suspended of deposited sediments, raising the need to dredge existing ponds or construct additional sites nearby (Foster, 2011). Pond lifespan is related to surrounding land use and land management practices, sediment load of inflows, drainage area, and pond size. Ponds initially constructed in the 19th century had much shorter lifespans than those constructed in the mid-1900s due to improved agricultural management practices that reduced erosion and sediment transport and introduction of technologies for extending the useful life of impoundments (Fairchild *et al.*, 2012). As did Buddemeier (2005), we identified many ponds which appeared to fill with sediment quite rapidly, sometimes within a decade or two. Others did not receive the sustaining flow that was perhaps anticipated prior to their construction and rarely or never hold water except after the most extreme events. On the other hand, a number of ponds appeared to be little changed after more than 70 years in the watershed.

The most common reason for pond failure is construction on unsuitable soils, both for the pond bottom and embankment itself (Tuttle, 2003). Even in cases where the dam does not catastrophically fail itself, inefficient function of the pond may lead landowners to breach the structure and abandon the site. The majority of ponds in the watersheds we examined are located on soil types that present moderate to severe challenges to pond construction, largely due to seepage concerns (Allison, 1991, Clower, 1980). Areas with widespread limestone are poor choices for pond sites, offering one reason why ponds in certain portions of our study area appear chronically dry. Geology simply does not support water retention

In addition to effects on sediment transport, farm ponds have the potential to alter local hydrological processes. While cumulative surface area of ponds is a very small proportion of total land area across large landscapes, in many cases they represent water resources that were otherwise entirely absence prior to European settlement (Buddemeier, 2005). In our survey, almost all identified ponds were located along natural drainage pathways. Very few were in flat uplands away from areas of channelized flow. On-channel sites typically are the first occupied during periods of pond construction (Fairchild *et al.*, 2012). As a result, the trap efficiency of these ponds and ability to retain inflows is likely much greater than if the ponds mostly occupied sites characterized by sheetflow processes. Logically, ponds sites are chosen for their potential to retain maximum water volume and provide a relatively dependable water source during dry periods. However, this retention also increases the residence time of water and raises the potential for localized deep drainage and groundwater recharge. At least one study of a mixed agricultural-forested watershed suggested the possibility for increasing local groundwater storage as a result of retention in small ponds (Juszczak *et* *al.*, 2007). However, in a study of regional processes in Oklahoma, proliferation of farm ponds was identified as possibly leading to reductions in streamflow and peak flow (Esralew & Lewis, 2010). Similar results were also revealed in studies of watersheds in other locations in Texas (R. J. Brandes Company, 2011) and Alabama (Chaney *et al.*, 2012), though streamflow effects appear to diminish in regions with higher annual precipitation. Though the effect of any single small pond on watershed yield is infinitesimal, the cumulative effect of these features across the landscape may affect downstream water use. Though described in effect as "small," water storage in rangeland ponds in Arizona were shown to account for approximately 6% of total annual watershed streamflow (Milne & Young, 1989).

Small ponds also play an important ecological role where they are present, with both positive and negative consequences. In addition to their hydrogeological impacts, small farm ponds can serve as critical refugia in certain systems and serve as local sinks for organic matter, but they may also contribute to significant fragmentation of riparian systems caused by their high density in certain locations. This interruption of multiple flow pathways has significant impacts on watershed-scale biogeochemical processes and organism distribution (Brainard & Fairchild, 2012, Chin *et al.*, 2008, Downing *et al.*, 2006). Even the smallest of farm ponds can have negative ecological effects on stream corridors (Tuttle, 2003). An interesting observation by Buddemeier (2005) notes the rapid installation of small ponds compared with the time horizons used for modeling ecological, climatic, and hydrologic trends. As a result, efforts to model large-scale or even local processes and estimate biogeochemical budgets might be severely hampered without the incorporation of these important elements.

It remains to be seen whether ponds have proliferated as a direct result of land use change, increased human demands on the water resources of the landscape, increased prioritization of conservation practices, or changes in funding for and management practices. A chronological inventory of these variables in areas experiencing increases or stable levels of cropland area would provide a helpful point of comparison. Regardless of causation, it is very likely that trends in both cropland area and ponds are related by the local relative scarcity of water, particularly in dry periods. Both pond construction and cropland abandonment have been greatly reduced in recent years. This is probably due to the fact the vast majority of cropland has already been lost and most favorable pond sites are already occupied. There is little cropland in production that still may undergo conversion and few locations for ponds where they currently do not exist. Lands that do remain in cultivation generally are streamside fields on alluvial soils. These sites are ideally suited to dryland agriculture and conceivably may be in production for decades to come, making them less likely to experience pond installation. Yet if demand for additional pond sites increases, greater competition may develop between pond and agricultural production uses for the diminishing area available.

The Lampasas Cut Plain has some of the highest densities of small ponds in the nation. Though it requires high inputs, the strength of an intensive visual assessment is made clear by the more complete information it provides, identifying sites that might be missed by automated approaches and providing information on pond maintenance activities. Overall, the high abundance of the ponds demonstrates the effect of human actions not just on the water cycle and geological processes of local waterways but on the whole of the landscape itself. It truly is a fundamental transformation of entire watersheds.

CHAPTER V

WATERSHED-SCALE EFFECTS OF LAND COVER DYNAMICS ON EROSION AND HYRDOLOGY IN CENTRAL TEXAS RANGELANDS

Introduction

Woody plant encroachment is a widespread phenomenon in rangelands around the world with number of proposed causes, including land use change, fire suppression, overgrazing by livestock, and enrichment of atmospheric carbon dioxide (Archer *et al.*, 2011). While these different drivers may play varying roles in differing landscape contexts, there is also a great deal of uncertainty about the ecological effects of increases in shrubs and trees on landscapes where they were not present historically. Shrub encroachment often has been implicated as reducing rangeland water yield and thus downstream water supplies (Hibbert, 1983). Supposed increases in water yield stemming from control of woody plants have even been suggested as a mechanism for luring investment from urban centers to rural areas (Griffin & McCarl, 1989, Rothe & Raabe, 2000). As a result of these and numerous other motivations, widespread management efforts have been conducted in places where woody plant encroachment has been observed. However, hard evidence of large-scale effects of either encroachment or removal are scarce. In reality, encroachment by woody plants can have a variety of hydrological effects, and the precise response is dictated by local geography, climate, and runoff mechanisms (Huxman *et al.*, 2005). Looking at the relative contribution of baseflow to streamflow over time sheds light on local runoff processes and is a reliable means of understanding the effects of vegetation change on local water budgets.

The potential for soil erosion is determined by a number of interacting factors, including land use, plant cover and productivity, soil attributes, local precipitation characteristics, and climatic trends (Nearing, 2001). Woody plant encroachment has been associated with ecosystem degradation and increased rates of erosion (Grover & Musick, 1990, Huxman *et al.*, 2005). Yet very few studies exist on the change in sediment dynamics corresponding with brush management, particularly at the watershed scale (Archer *et al.*, 2011). At the intersection of hydrology and sediment dynamics, and compounding future water resource needs, storage capacity of existing reservoirs in Texas is projected to decrease by 18% - over 3 million acre-feet – largely due to sedimentation by 2060 (Texas Water Development Board, 2007). Thus understanding rangeland processes as they relate to water and sediment have supreme importance, both for the rangelands themselves and their effects on downstream areas.

Late in the first half of the 20th century, federal programs began constructing small dams on upper portions of tributary streams across the nation to improve flood control as a result of the Flood Control Act of 1944, the Pilot Watershed Program, and the Watershed Protection and Flood Prevention Act of 1954 (Dunbar *et al.*, 2010). These structures were installed in an effort to mitigate downstream flooding during extreme rainfall events by detaining flows high in drainage basins and gradually releasing stored water volume over time. Initially implemented by the Soil Conservation Service (now USDA-NRCS) for flood mitigation, these 11,000 flood control structures (FCSs) have come to provide additional benefits many locations, including water supply, fish and wildlife habitat, and recreation opportunities (Bennett *et al.*, 2002, Caldwell, 1999, Van Liew *et al.*, 2003). These benefits have continued well beyond the 50-year design life of the dams. Combined, Texas and Oklahoma boast approximately 40% of these FCS sites (Dunbar *et al.*, 2010), with 34 present in the Lampasas River basin.

These dams face many challenges to their continued function including physical and structural deterioration, encroachment by urban and suburban development, and loss of storage capacity due to sedimentation. Increasing numbers are being dredged or decommissioned to maintain usefulness or remove potentially significant hazards to downstream areas (Dunbar *et al.*, 2010). However, as each FCS and its resulting reservoir serve to detain water flows and trap sediment transported from upstream portions of a watershed, these sites have the potential to provide a great deal of information on the history of watershed dynamics in the local ecosystem by acting as archives of past conditions. This is particularly true in locations that have not experienced dam failure or spillway overtopping. When FCS sediments can be removed and examined for chronological markers, they provide a detailed sequence of hydrological trends and sedimentation events in the watershed, similar to observing tree rings to interpret life histories in certain ecosystems. Cesium-137 is a product of nuclear fission and thus is present in the environment only as a result of atomic weapons testing and accidental emissions from nuclear power generation facilities (Ritchie & McHenry, 1990). As a result, this isotope makes an ideal tracer for chronologically sequencing, as peak weapons testing occurred in 1963, with a smaller peak sometimes present corresponding to the Chernobyl incident in 1986. Cesium-137 also preferentially adheres to soil clay particles, which are usually the most easily eroded from the landscape. As explained by Bennett *et al*. (2002) distinguishing between preimpoundment surface material and sediment deposited after FCS impoundment is crucial to establishing sedimentation rates. Relying on sediment texture is not always reliable. However, the incorporation of additional tracers beyond cesium-137, such as lead-210, provides additional to discriminate soil from sediment, with the original land surface depleted of the radioisotope. Application of these analyses allows objective dating of reservoir sediments.

Climate and parent material are the natural drivers of ecosystem structure and function, but anthropogenic land use is often just as important (Burke *et al.*, 1994). By linking sediment and hydrological response to watershed changes associated with changes in land use and land cover, it becomes possible to quantify the relationship between range condition and ecosystem services.

Our objectives are to 1) estimate changing watershed sediment yields using core analyses of small impoundments and 2) identify changes in hydrological response over time in dynamic rangelands with long-term streamflow data. We anticipate documenting a gradual decline in sediment yields from watersheds due to improving rangeland conditions. In addition, we hypothesize that local stream baseflows generally have been enhanced as a result of increased infiltration of precipitation into the soil surface and underlying horizons.

Materials and Methods

Location Description

To conduct analyses of the dynamics of sediment and hydrology in changing rangeland watersheds, we selected eight FCS reservoirs in Lampasas and Mills Counties in central Texas (Figure 5.1). These sites were chosen based landscape position, year of impoundment, and availability of access to the private lands on which they are located. Older sites were preferred to utilize cesium-137 markers. All reservoirs and their watersheds fall within the Lampasas Cut Plain, a transitional zone between the Edwards Plateau, the Cross Timbers, and the prairies of north Texas. Sites in Lampasas County are located within the Sulphur Creek drainage of the Lampasas River and were constructed between 1958 and 1961. Lampasas Site X is downstream of Lampasas Sites 1-3, which are in parallel with one another. Those in the Bennett Creek drainage of Mills County are somewhat younger, completed in 1974. All reservoirs and their watersheds fall within the Lampasas Cut Plain, a transitional zone between the Edwards Plateau, the Cross Timbers, and the prairies of north Texas (Allison, 1991, Clower, 1980).The drainage area of watersheds contributing to the selected reservoirs ranges vary in size from 1.24 km^2 to 50.92 km^2, with a median of 27.74 km^2. Average annual precipitation ranges from 710 mm in Mills County to near 780 mm in southernmost portions of watersheds in Lampasas County. Rainfall is somewhat evenly distributed, with a slight peak in spring. Lowest elevations of 322 m are found within Lampasas city limits, with maximum elevations to 536 m in central Mills County. Prevailing geology is Cretaceous limestone. Soils across the area generally are loamy and clayey, with sandy loams present in far northern reaches approaching the Cross Timbers. All soils are susceptible to sheet and gully erosion. Basic characteristics for the reservoirs surveyed in this study are located in Table 5.1.

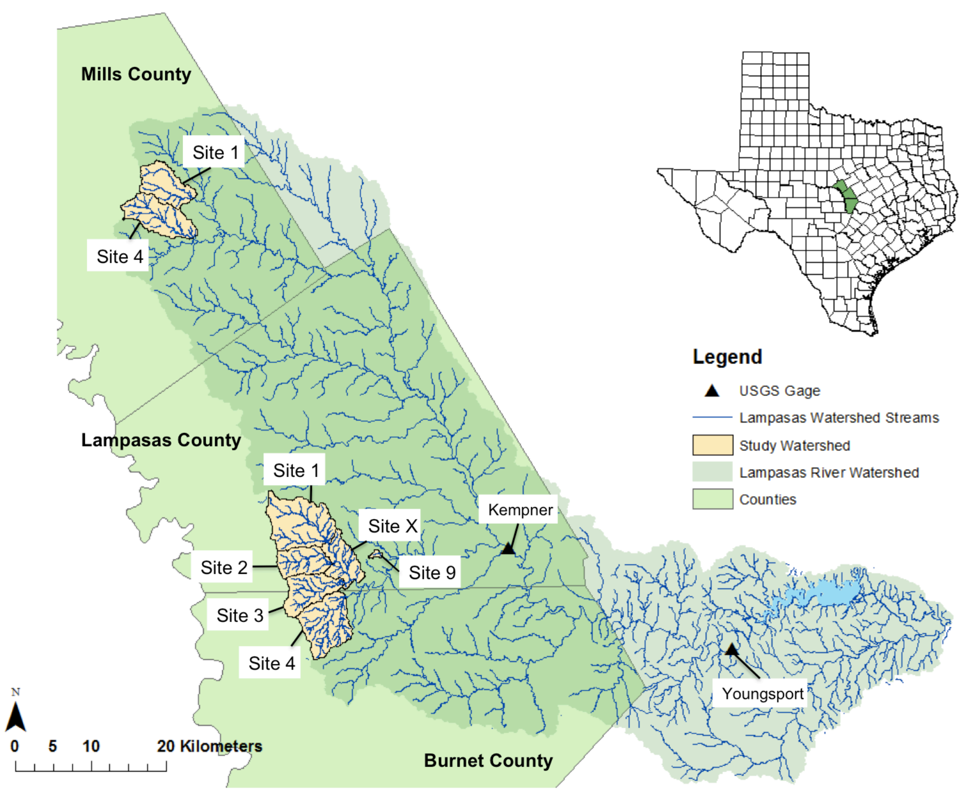


Figure 5.1. Map of study watershed location with USGS gage sites.

Table 5.1. Description of basic reservoir characteristics.

FCS Designation	Primary Inflow	Year Constructed	Surface Area (km^2)	Watershed Area (km^2)
Lampasas Site 1	Donalson Creek	1959	0.20	50.92
Lampasas Site 2	Pitt Creek	1959	0.18	23.23
Lampasas Site 3	Espy Branch	1958	0.11	27.49
Lampasas Site 4	Pillar Bluff Creek	1960	0.07	41.20
Lampasas Site 9	Cemetery Creek	1960	0.02	1.24
Lampasas Site X	Bean Creek	1961	0.20	23.11
Mills Site 1	Middle Bennett Creek	1974	0.14	34.61
Mills Site 4	Mustang Creek	1974	0.15	27.98

Estimating Watershed Sediment Flux

To explore the effect of changing land use change and vegetation cover, we obtained sediments from the main body each reservoir, near the dam structure and along areas corresponding with the pre-impoundment floodplain as determined by exploratory hydroacoustic surveys. This approach reduces the likelihood of capturing highly mixed sediment profiles, which occur frequently near stream inflows. It also ensures the collection of very fine sediments that remain suspended for longer periods of time before settling from the water column, and it is these smaller particles to which the radioisotopes more effectively adhere (Bennett *et al.*, 2002). We extracted cores using a portable vibracoring system, from Specialty Devices, Inc., suspended from a modified pontoon boat, when possible with adequate reservoir depths. This device is designed for capturing unconsolidated sediments in saturated conditions with minimal disturbance and compaction (Lanesky *et al.*, 1979). In cases when water levels did not support the launching or maneuvering of a large boat, we used two small fishing boats, joined with a stabilizing A-frame, to support the coring apparatus. Cores were removed using a 7.62 cm (3-inch) aluminum irrigation pipe lowered to the point of refusal in the reservoir bottom, ideally penetrating the uppermost portion of the preimpoundment surface. Cores were sealed and transported upright to cold storage (~5°C). A subset of cores was split vertically to allow examination of physical characteristics and presence of plant roots or detritus for interpreting site history.

To prepare sediment cores for radioisotope analysis, we sectioned cores laterally in 3 cm intervals and followed protocols recommended by IAEA (2003). Sections were weighed and then dried in an oven at 100°C until constant weight was obtained (approximately 48 hours). After determining dry weight, resulting hard aggregates were ground and homogenized using a BICO, Inc. soil pulverizer before passing through a 2 mm sieve. A subsection of each homogenized sediment interval was sealed in a 50 mm x 9 mm Petri dish and shipped to the Institute for Geophysics, Jackson School of Geosciences at the University of Texas at Austin for radioisotope analysis. Counts for lead-210 and cesium-137 were performed using high-purity germanium (HPGe) low-level gamma detectors.

Following radioisotope counts, we plotted each radioisotope with depth. For cesium-137, we identified the depth of peak activity and computed linear sedimentation rate for periods prior to and after 1963 for each watershed. Using lead-210, we calculated linear sedimentation rates for each core according to (Robbins, 1984), identifying periods of change over the depth of the sample. Linear sedimentation rates were normalized by watershed area.

Hydrological Response

To investigate hydrologic response to changing land cover conditions, we obtained historical data for each streamflow and precipitation. Precipitation data are available for numerous stations throughout the Lampasas River basin since 1895. USGS stream gage data pertinent to this study are available from two different stations, which overlapped for a portion of their temporal coverage. The gage at Youngsport (08104000) operated between 1924 and 1980 with a drainage area of 3,212 km^2, while the site at Kempner (08103800) has remained active from 1980 to the present, receiving runoff from a drainage area of 2,119 km^2.

Comprehensive rainfall data were assembled from weather stations in the watershed, with annual precipitation totals derived using a hybrid Thiessen polygon approach weighted by area falling under each of three different long-term precipitation zones. We assessed the relationship between the Kempner and Youngsport stream gage stations by plotting annual streamflow at the sites over the period in which both were active. Results are shown in Figure 5.2. Kempner streamflow as a proportion of Youngsport streamflow fluctuated between approximately 0.5 and 1.3 from 1963 to 1980, averaging approximately 0.9 of that of the lower station. This value was assigned as a correction coefficient to construct a continuous record incorporating data from both stream gage locations from 1924 to 2010.

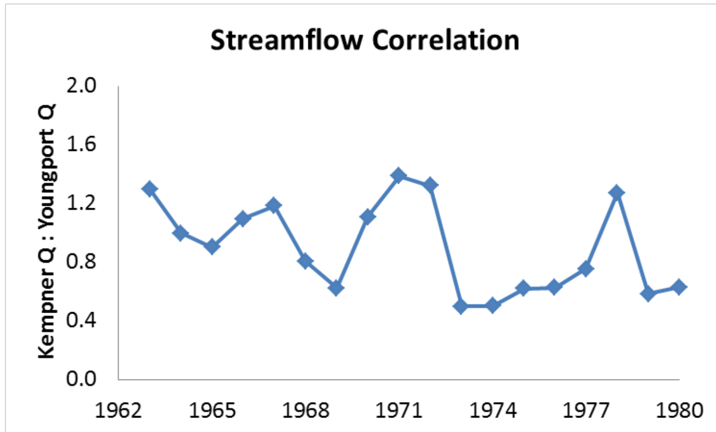


Figure 5.2. Correlation of annual streamflow at Kemper (08103800) and Youngsport (08104000) streamflow gage stations.

After developing a long-term annual streamflow dataset for this portion of the Lampasas River, we performed a baseflow separation to quantify the relative contributions of baseflow and stormflow over time (Arnold & Allen, 1999, Arnold *et al.*, 1995). This procedure is objective and reproducible and sheds light on water budget components by separating slower-responding groundwater and quick-responding stormflow, which can be interpreted in comparison with dynamic watershed land use and land cover conditions. Directional long-term changes were detected using a nonparametric Mann-Kendall trend test, a common approach in identifying hydrological trends (Lettenmaier *et al.*, 1994, Salas, 1993). Tests were performed using a two-tailed test with a = 0.10 to determine significance. Trends in annual precipitation were evaluated with a similar approach using the composite record for the area.

Results

Estimating Watershed Sediment Flux

Sediment core depths varied widely, from greater than 150 cm in Lampasas Site 1 to little more than 1 cm in Lampasas Site X. Peaks associated with cesium-137 are visible only in Lampasas Site 1 and Lampasas Site 2, with the falling limb of a similar peak visible in Lampasas Site 3 (Figure 5.3). Lampasas Site 4, Lampasas Site X, and Mills Site 4 produced relatively shallow sediment cores that do not exhibit a pronounced cesium-137 peak. Comparison of linear sedimentation rates from some different cores is problematic due to collection of incomplete cores that did not achieve the preimpoundment surface. In these cases, only a minimum rate could be computed. With available data, it is apparent that sediment yield decreased at both Lampasas Site 1 (from >0.736 mm/yr/km^2 to 0.558 mm/yr/km^2) and Lampasas Site 2 (0.969 mm/yr/km^2 to 0.344 mm/yr/km^2) when comparing conditions before and after the 1963 cesium-137 peak. In terms of geographical comparison, Lampasas Site 1 has displayed a greater sediment yield than adjacent Lampasas Site 2 in recent years, the downstream Lampasas Site X has experienced a much lower sedimentation rate than any other area documented here. A summary of sedimentation rates using the different approaches is found in Table 5.2.

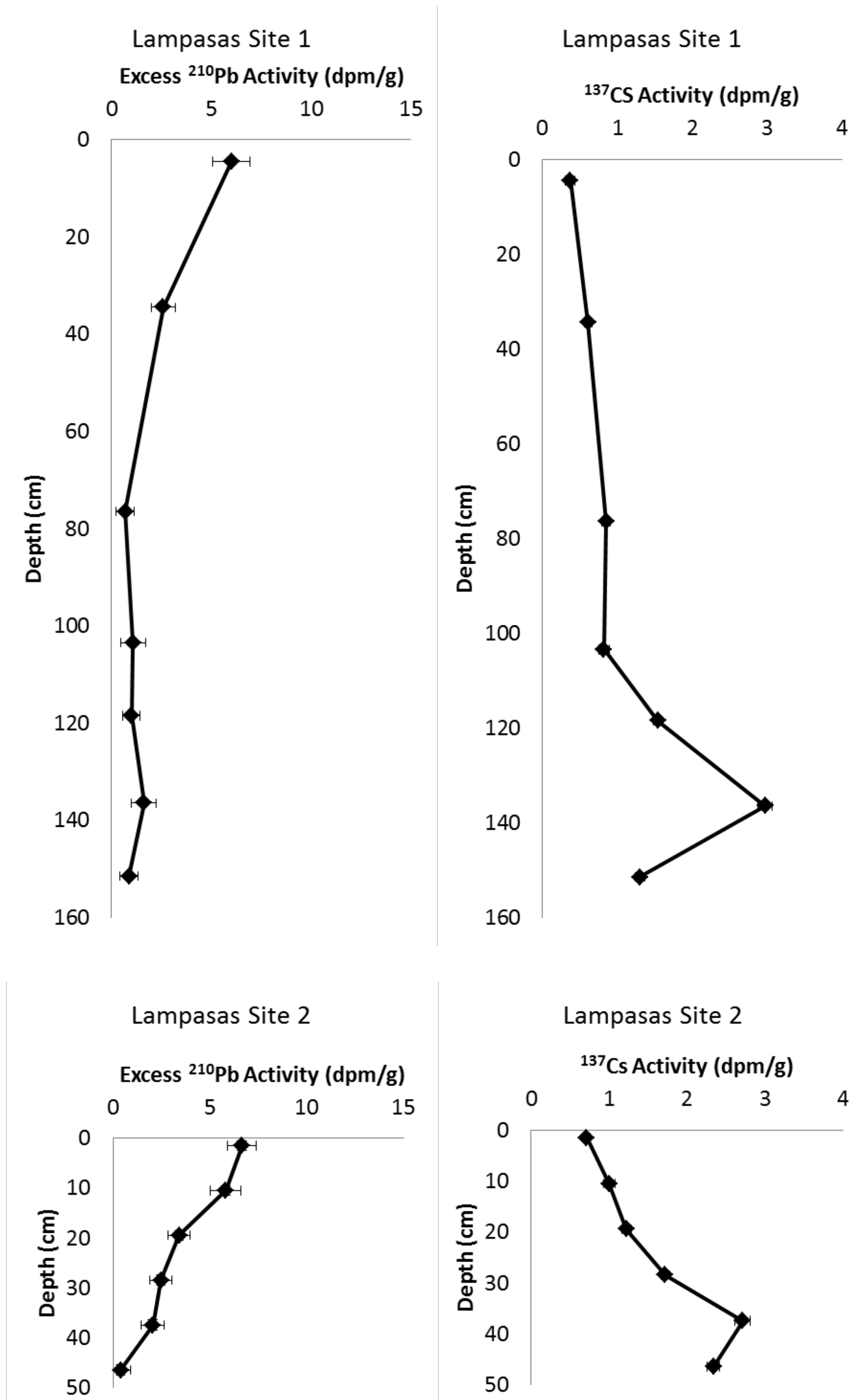


Figure 5.3. Sediment radioisotope profiles for selected cores from Lampasas Sites 1-4, Lampasas Site X, and Mills Site 4.

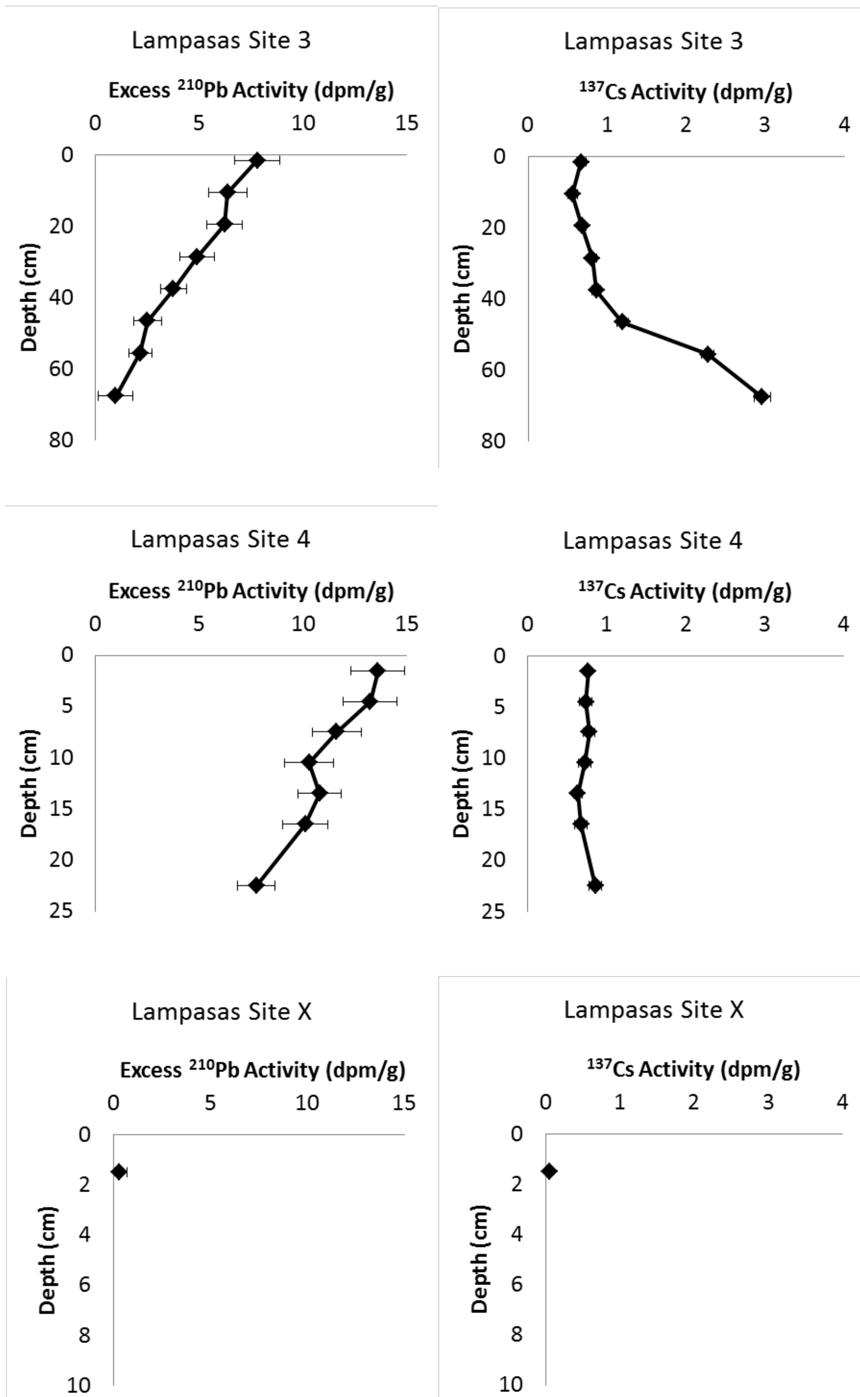


Figure 5.3 Continued.

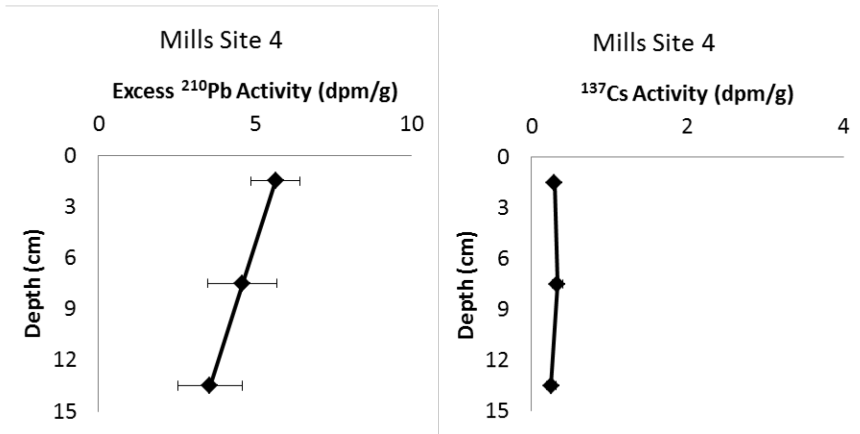


Figure 5.3 Continued.

Table 5.2. Linear sedimentation rate estimation by cesium-137 peak identification.

Core Source	Sediment Depth (cm)	^{137}Cs pre-1963 LSR (mm/yr/km^2)	^{137}Cs post-1963 LSR (mm/yr/km^2)
Lampasas Site 1*	> 153	> 0.736	0.558
Lampasas Site 2	48	0.969	0.344
Lampasas Site 3*	> 72	-	> 0.472
Lampasas Site 4*	> 27	-	-
Lampasas Site 9	N/A	-	-
Lampasas Site X	3	-	0.012
Mills Site 1	N/A	-	-
Mills Site 4*	> 15	-	> 0.129

*Certain cores were not a complete record of deposited, as excess lead-210 indicated the preimpoundment surface was not reached. For these cores, linear sedimentation rates given are minimal values.

Hydrological Response

While there is a great deal of interannual variability in stream gage data, there is no apparent long-term trend in baseflow in the Lampasas River below the watersheds examined here. In fact, baseflow appears almost unchanged over the period 1935-2010 (Figure 5.4). Similarly, other measures of local hydrology largely appear to demonstrate no change when considered over multiple decades. For instance, total annual total precipitation has remained nearly constant over the same timeframe despite large fluctuations from year to year, averaging approximately 800 mm (Figure 5.5). In fact, the rainfall regime in this area has remained unchanged since the 19th century. Furthermore, runoff (streamflow) as a proportion of rainfall at the basin scale has gone unchanged at approximately 0.08 (Figure 5.6).

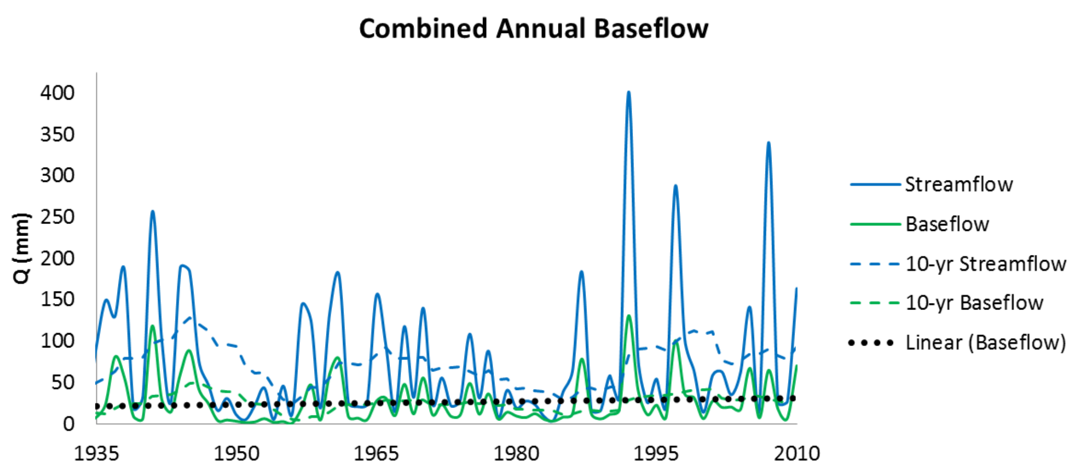


Figure 5.4. Historical baseflow separation of Lampasas River, 1935-2010. Baseflow is unchanged.

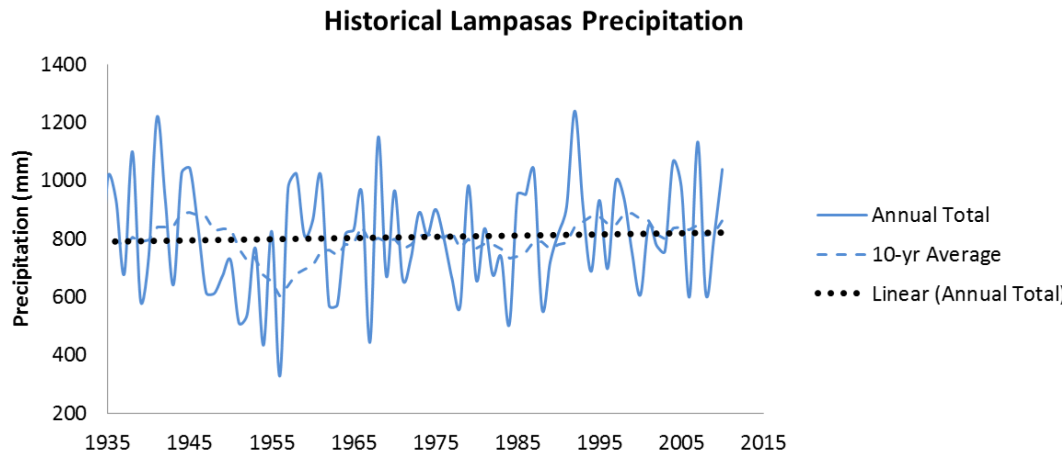


Figure 5.5. Historical precipitation trends, 1935-2010. Total annual precipitation is unchanged.

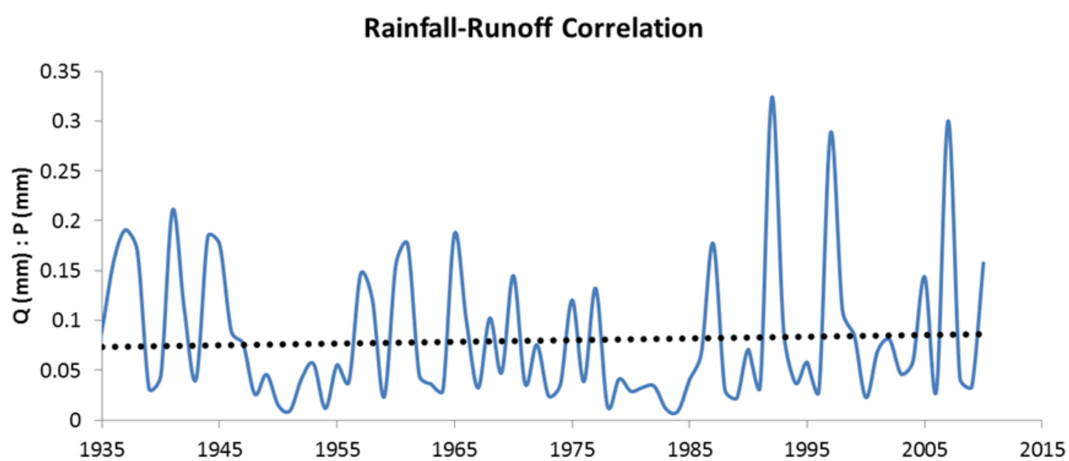


Figure 5.6. Historical relationship between total annual streamflow and total annual rainfall, 1935-2010. Streamflow has averaged approximately 8% of total rainfall over the entire period.

When baseflow is considered as a proportion of total streamflow, there does appear to be a slight increase over time (Figure 5.7). Interestingly, this trend is apparently driven by streamflow data toward the end of the study period, which are marked by several years with very high total streamflow.

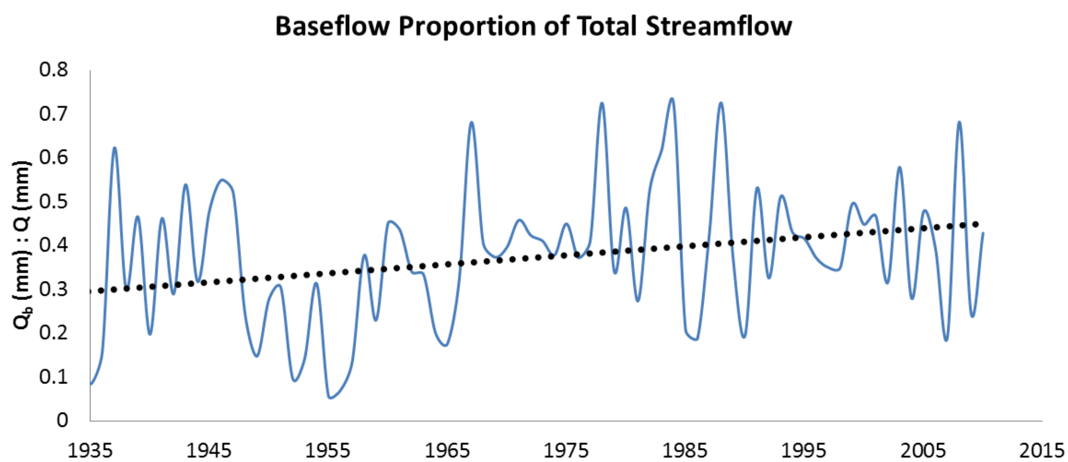


Figure 5.7. Historical relationship between baseflow and total streamflow, 1935-2010. A small increase is apparent.

Discussion

Examining streamflows and sediment dynamics provides a wealth of information on rangeland health and function, in addition to highlighting changes in landscape processes over time. Linking responses of water and sediment transport to ecological changes improves our ability to understand the effects of land management and plan for future conditions. This portion of the Lampasas Cut Plain has experienced widespread woody plant encroachment after a prolonged period of decline, with woody plant cover in recent years returning to levels of the 1940s. In this context, evidence of no or minimal long-term change in water or sediment yield is just as valuable as documentation of major shifts in watershed function.

From the sediment results yielded by analysis of cesium-137 from FCS reservoir cores, it appears that there are both spatial and temporal causes of variation in sediment dynamics in the area. Among the cores we have examined, cumulative linear sedimentation rates prior to 1963 exceeded those from 1963 to the present. Over this period, there has been both a decrease in total cropland area in favor of rangeland use across all portions of the study and an increase in woody plant cover among the watersheds in Lampasas County. Both of these trends may play a role in soil erosion. As early as 1860, Lampasas and Burnet Counties had become major players in Texas wheat production, signaling a relatively long tenure of cultivation on the Lampasas Cut Plain (Jordan-Bychkov *et al.*, 1984). Conversion to cropland cultivation often affects soil aggregates negatively and depletes accumulated organic matter, making erosional losses more likely (Burke *et al.*, 1994). As in other areas, significant erosion losses lead to a depression of productivity in the short term and cropland abandonment in the long run (Brown, 1978). By 1930, cropland in the Lampasas Cut Plain started to decline, and cultivated fields were abandoned and converted to use as rangelands (Ramankutty & Foley, 1999b). While the abandonment of cultivation may support renewed increases in soil organic matter, the effects of compaction on soil properties may be much more permanent (Hoshino *et al.*, 2009).

Though excessive overstocking of rangelands by cattle and sheep was rampant regionally over the second half of the 19th century, animal numbers had decreased significantly by the middle of the 20th century (Bentley, 1898, Box, 1967). However, with the transition from cropland to rangeland, animal number again increased to levels not seen for decades (Wilcox *et al.*, 2012). Grazing and reduced productivity due to plant stress cause declines in soil organic matter, leading to greater susceptibility to erosion and higher sediment yield (Diodato, 2006, Smith *et al.*, 1984). It was recognized very early on that resting central Texas rangelands, particularly during drought, offered an opportunity to maintain the long-term capacity of grazing lands, though this was rarely done in practice (Bentley, 1898) and remains uncommon today due to economic considerations. This pressure from livestock grazing likely contributed to watershed sediment yields to some degree, though animal numbers have been somewhat constant over the last four decades and cropland area has declined to very low levels and shows no sign of increase.

With the removal of regular disturbance from site preparation and harvest of cropped fields and due to the demands of providing open grazing lands for livestock, woody plant management efforts have been widespread in the Lampasas Cut Plain over the last century. Even in the absence of brush control, woody plant encroachment can increase soil erosion losses if encroachment results in a loss of herbaceous cover and a rise in the proportion of bare ground (Archer *et al.*, 2011). Once removal of juniper and other woody species commenced, initial mechanical methods of hand grubbing and root plowing and grubbing likely had a significant impact on local soils, increasing the potential for erosion (Hamilton & Hanselka, 2004). Even where juniper control has occurred, massive soil erosion sometimes results if removal efforts are not followed by proper grazing management (Nelle, 1997b). Thus, if agricultural activities are pursued beyond sustainable levels, erosion can occur both before and as a result of cropland abandonment. The same is true for woody plant cover. Erosion potential may increase with an expansion of shrub cover but may be even greater following brush management if great care is not taken following removal. Clearly there are many opportunities for land management practices to inadvertently elevate erosion rates. Indeed, sediment yield during non-drought conditions is generally linked to land use and management activities rather than runoff (Allen *et al.*, 2011).

The abundance of constructed small ponds on the landscape has likely mitigated the loss of sediment from watersheds we examined, since these features serve as numerous sinks for eroded soil on the landscape. Pond density is greatest in the central and southeastern part of the country, and it is in the agriculture-dominated portions of these regions that most of the sediment accumulation in impoundments is concentrated (Renwick *et al.*, 2005b). The combined trends of prior agricultural expansion and impoundment construction serve to simultaneously increase both edge-of-field sediment yield across the landscape and also internal sediment storage. As a result sediment is retained in upstream portions of watersheds while being withheld from delivery to downstream areas. Indeed, studies have shown that much of the sediment lost from upland areas is still on the landscape in upland, colluvial, and alluvial deposits and in impoundments (Meade, 1982). In many instances, comparatively little eroded material is exported from agricultural though it is transported internally. Over time, deposition in these sinks, particularly along lower-order streams, positions them to be remobilized at a later time, becoming sources of sediment in the long run. As surface impoundments have become more common, they have grown in importance as sediment sinks relative to alluvial and colluvial deposition. Renwick *et al.* (2005b) found that more than 21% of land area in the conterminous United States is upstream from at least one impoundment, meaning nearly a quarter of the potential sediment source area in the country contributes to a pond or reservoir. In our study area, this number likely is even higher, with such a high density acting to shrink the effective catchment size of the FCS reservoirs over time and reduce the area of upstream areas actively delivering sediment to the sites from which we collected sediment cores. As a result, impoundments present a significant challenge to predicting sediment yield at increasing watershed scales. According to Renwick et al. (2005b), at least as much sediment has accumulated in impoundments of all sizes as has been lost through sheet and rill erosion from uplands, and a major component of sedimentation processes now occurs in submerged environments. Additionally, deposition in these sinks is likely to lead to much longer residence times than sediments laid down in alluvial or colluvial deposits and those suspended in the water column of flowing streams. Even among the FCS reservoirs we sampled, some potential coring sites were too dry or overburdened with sediment to permit coring. Thus, there may have been some selection bias in identifying coring locations, leading to the reservoirs with highest sedimentation rates being excluded from investigation due to lack of accessibility. Considering these unsampled sites, there is a great deal of information still on the landscape, potentially including documentation of historical sedimentation impacts even greater than those sites we did examine.

It is important to note that Lampasas Sites 1-3 each exhibited much higher linear sedimentation rates than did Lampasas Site X. Lampasas Site X received runoff and sediment loads from the upper watersheds prior to the impoundment of associated reservoirs in 1958-1959. The sediment that would have been delivered to the downstream reservoir is now retained higher in the tributary watersheds. While this observation indicates a great deal of sediment storage in these impoundments, it may also indicate a key local source of sediment. The watersheds of Lampasas Sites 2 and 3 and Lampasas Site X are of approximately the same size, yet the former have experienced linear sedimentation rates more than 30 times greater than the latter. Soils in both areas are similar. One of the key differences between these areas is streamflow. Since nearly all runoff from the upstream watersheds is retained by FCS reservoirs, downstream runoff is much lower, only receiving in-channel flow during peak rainfall events. With comparable watershed sizes and soils yet differing streamflows, it appears likely that a large source of sediment in this landscape is channel erosion, which has been observed elsewhere (Bartley *et al.*, 2007, Mukundan *et al.*, 2011, Renwick *et al.*, 2005a, Wasson *et al.*, 1998). In this scenario, previously deposited materials are remobilized and are transported downstream. Even when upland erosion has slowed, downstream sedimentation may very well continue at elevated rates as previously eroded sediments are gradually transported down the watershed. In fact, where large alluvial deposits exist resulting from accelerated erosion of upland soils, high sediment yields may continue for a very long time. As ponds impoundments have multiplied across the landscape, it is likely that an increasing proportion of these deposits is being retained locally due to widespread manipulation of the sediment transport system. It is probable that a significant portion of this stored sediment results from agricultural activity when crop production and livestock numbers were much higher. Without regular runoff volumes to support channel and bank erosion, Lampasas Site X experiences much lower rates of sedimentation.

In these watersheds fed by intermittent streams that typically flow only after rainfall events, characteristics of climate and precipitation play an important role in soil loss and sediment transport. The recurrence interval of severe drought combined with intense storms to mobilize sediment is an important factor in realizing potential losses. Dunbar et al. (2010) found that under most recent conditions, FCS reservoirs may retain sediment storage capacity for many centuries, while a return to prolonged severe drought such as that of the 1950s combined with a few powerful rainfall events may lead to complete sedimentation of the same sites in only s few years. In general, more sediment is generated by a single storm during drought conditions than by an equivalent storm during wetter conditions, but total sediment transport in drought generally is lower due to the scarcity of intermediate flows (Allen *et al.*, 2011, Kochel *et al.*, 1997). However, Dunbar et al. (2010) found the opposite effect during extreme drought in central Texas. Drought has been a regular and recurring condition in the Lampasas Cut Plain, with important drought events in the 1950s and again after 2007 (Figure 5.8). When these conditions combine with high flow events in the Lampasas River basin, the potential for erosion is very high. Significant flood events followed the drought of the 1950s as well as those of the early 21st century (Figure 5.9). The watersheds investigated here are roughly equidistant between watershed sediment studies conducted near Brady to the west (Dunbar *et al.*, 2010) and Riesel to the east (Allen *et al.*, 2011). Reservoir sediments in the westernmost study were found to contain 80% drought-derived deposition, compared with only 9% of watershed yield near Riesel. Differences are likely due to variation in local drought intensity, inherent climatic differences and number of storm events, and plant cover characteristics. In the drier landscape of Walnut Gulch, Arizona, Nearing et al. (2007) found a mere 6-10 rainstorms produced half of the sediment from nearly all watersheds studied.

The greater ability of individual storms to produce sediment during drought and naturally dry climates is related to reduced plant cover as well as to lower antecedent soil moisture (Dunbar *et al.*, 2010). In dry conditions, infrequent precipitation events combine with lower levels of vegetation and greater extent of bare soil as erosion is less common yet of greater magnitude when it does occur. Clay-dominated soils show marked increases in erodibility during drying conditions that cause cracking and the development of rills (Kuhn & Bryan, 2004). The relative importance of drought severity, soil erodibility, plant cover, and precipitation intensity is key in predicting the nature of drought sediment yield from a specific location.

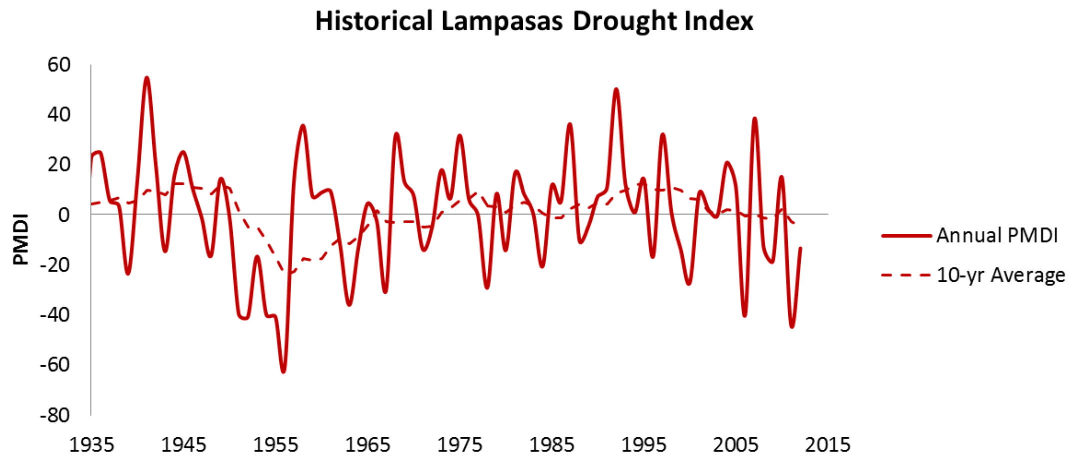
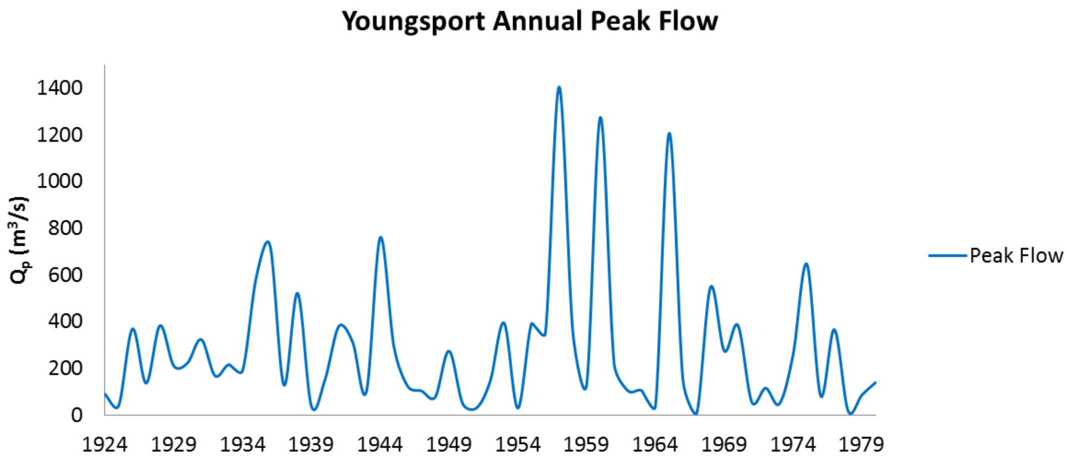


Figure 5.8. Historical Palmer Modified Drought Severity Index (PDMI), 1935-2012.



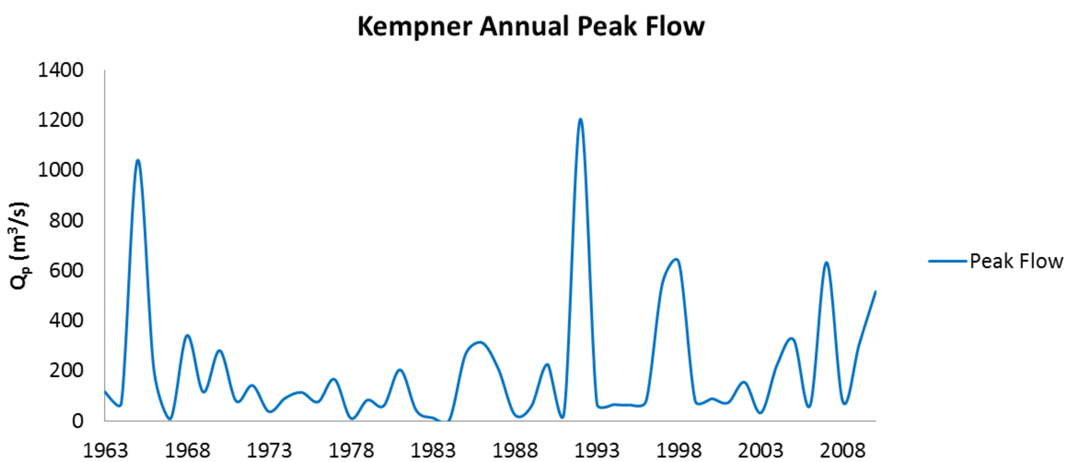


Figure 5.9. Historical peak flows for Youngsport (08104000) and Kempner (08103800) streamflow gaging stations, presented in maximum annual discharge.

There is a great deal of difficulty in extrapolating linear sedimentation rate to rangeland erosion rates due to the great extent of internal storage in uplands, alluvial deposits, and small ponds. It would be more appropriate to consider reservoir deposition as an indicator of sediment yield on a watershed basis, understanding that the number of sinks on the landscape is large and that time lags between erosion and downstream deposition increase with spatial scale. Sediment flux demonstrates scaling effects, with delivery ratio and linear sedimentation rates generally decreasing with an increase in basin size (Renwick *et al.*, 2005b, Smith *et al.*, 2002). Very high levels of erosion may exhaust the landscape of the most easily erodible materials, leading to lower sediment yields in subsequent periods (Dunbar *et al.*, 2010). It is possible that this explains a portion of the decrease in linear sedimentation rates since 1963, but it is likely that this is an artifact of the sampling period. The years prior to 1963 were marked by historic drought conditions followed by major floods, conditions that are ideal for soil loss and sediment transport. It is probable that these same conditions caused very high rates of sedimentation during recent drought conditions, but increasing rates where masked when averaged with lower levels of deposition in the intervening wetter years.

Given the approach taken in this study, there is no way to determine the rates of erosion and sediment yield prior to the impoundment of the FCS reservoirs. It is possible that a great deal of evidence regarding the health and function of the Lampasas Cut Plain landscape has already been transported downstream. At the same time, it is very likely that a large percentage of eroded material is still stored internally in these watersheds. Though periodic water storage is very low due to local precipitation patterns, a great deal of potential sediment storage remains in most reservoirs we sampled. Many FCS sites in other locations have experienced sedimentation rates much lower than expected, with sediment retention averaging only 30% despite nearing the end of design life, and this is especially true of those completed since the 1960s in rural landscapes (Dunbar *et al.*, 2010).

In this study, we do not see changes in total streamflow or baseflow during either the current period of widespread woody plant encroachment or during the preceding regional decrease of shrub cover. In contrast with studies showing either increases or decreases in water yield with increases in woody plant cover, we find stability in the watersheds examined. The Lampasas River watershed appears hydrologically resilient in the face of widespread significant change across the landscape.

There is a commonly held belief that increases in woody plant cover negatively affect local water budgets, ranging from minor alterations to major shifts in hydrological systems. Woody plants in semiarid regions generally extract soil water for a greater proportion of the year than do herbaceous species, leading to higher rates of transpiration (Huxman *et al.*, 2005, Kemp, 1983). However, large shifts in transpiration as a percentage of total evapotranspiration are less likely to occur following woody plant encroachment of wetter climates. Indeed, it has been suggested that where runoff is less than 10% of mean annual precipitation, afforestation should result in a complete loss of runoff (Farley *et al.*, 2005). While shrub cover increases in Lampasas County and Mills County certainly have not progressed to the point of complete closed canopies in all areas, there is no hint of this impending elimination of streamflow, despite the fact that runoff averages only 8% of annual precipitation. Additionally, woody plant encroachment of riparian areas has received a great deal of attention for potential increases in water yield. However, expansion of shrub cover in the Lampasas Cut Plain has occurred in upland areas with little potential for directly affecting streamflows (Wilcox *et al.*, 2006). Hibbert (1983) takes a more conditional approach and proposes that some increases in upland water yield may be possible in rangelands in cooler climates having at least 450 mm annual rainfall with that precipitation concentrated in winter months. These conditions do not describe the rangelands across the Lampasas Cut Plain or elsewhere in Texas. In fact, the author argues that the only mechanism to augment water yield in grasslands and shrublands is through increase the overland runoff component of local water budgets. In fact, surface runoff may be reduced due to the diminished storage capacity of the soil as a result of landscape degradation (Wilcox *et al.*, 2008). Considering shrub-dominated landscapes in Texas, Wilcox (2002) reaches similar conclusions as Hibbert (1983), placing a precipitation threshold at 500 mm. However, the author also points out that a rapid connection between surface runoff and subsurface pathways must be present, as in the case of karst landscapes with occasional springs, which may experience some additional flow. In this case, if local runoff occurs as overland flow, little effect from shrub removal should be expected. It stands to reason that if woody plant cover can cause declines in streamflow, brush management efforts can reverse these trends and lead to higher water yields. But while runoff may increase at smaller scales when many specific site criteria are met, there are questions regarding the scalability of these observations, and perceived increases in plot studies may not be realized across larger areas (Wilcox & Huang, 2010). Though shrub cover has changed dramatically in our study area and precipitation is well above the 500 mm deemed necessary for a hydrological response, we see no decrease in annual streamflows during periods of woody plant increase.

But neither we do observe the increases in streamflow shown to accompany woody plant encroachment in the context of improving range condition. This trend has resulted from reduced grazing intensity on karst landscapes in multiple river basins on the adjacent Edwards Plateau (Wilcox & Huang, 2010). In these landscapes immediately to the west of our study area, reduced stocking rates have led to improved soil health and vegetation cover, reducing overland runoff yet boosting infiltration (Wilcox *et al.*, 2008). Indeed, both baseflow and stormflow can be affected by plant cover characteristics, and rangeland condition may be a more important determinant of streamflow than woody plant cover by itself. But in contrast with the Edwards Plateau and drier regions farther west, livestock numbers in the Lampasas Cut Plain have increased since the early 20th century, driven by a near tripling of cattle numbers since 1900 (Wilcox *et al.*, 2012). It is possible that rangeland conditions have not improved significantly over the last several decades due to a continued high degree of grazing pressure. Yet streamflows showed remarkably little variation even before these changes in local agricultural preferences, so it is unlikely that these changes have played a major role in altering local hydrology.

Though increases in water yield through shrub control have been shown to be realistic in very few circumstances and that increases in juniper and other woody species may actually signal improving landscapes that yield more water, woody plant control efforts continue in the pursuit of boosting water supplies. Yet even where there may be some small-scale hydrological benefits of brush removal at the plot level, the consequences of removing juniper at large watershed scales are questionable, and it is unlikely that any significant increases in water yield will result (Kuhn *et al.*, 2007). But despite the lack of hydrological payoff, there may be important benefits to wildlife habitat and livestock grazing, so many of these efforts continue for a variety of reasons.

Even after intensive brush control efforts, ideal range management practices generally encourage a profusion of grasses that itself would intercept and transpire a large percentage of precipitation, rendering potential water gains minimal at best on most soils (Nelle, 1997b). Clearly, there are many obstacles to realistic expectations of increased watershed yield following woody plant removal.

In addition to widespread changes in woody plant cover, small farm ponds have proliferated in this landscape since the beginning of the 20th century. The net hydrological effect of these numerous features is unclear. While they may serve to affect local groundwater interactions and increase recharge, they also increase local evaporation of impounded surface waters, especially during summer heating of shallow systems. This removes water volumes that would otherwise participate in regional water budgets as streamflow (Smith *et al.*, 2002). It is possible that these processes occur in tandem and alter the distribution of water flows within a watershed, as increased residence due to retention boosts recharge in uplands as an increasing proportion of rainfall is lost due to evaporation. Both mechanisms serve to redistribute water preferentially to upland areas relative to preimpoundment conditions favoring flows to downstream areas. In certain systems, small ponds have been suggested as leading to decreases in streamflow and peak flow by retaining high volumes of water throughout the year, especially during and after storm events (Esralew & Lewis, 2010). Juszczak *et* *al.* (2007) found localized groundwater recharge a possible effect of increased retention in small ponds. The increased contribution to groundwater far exceeded the surface storage capacity of the pond itself on an annual basis.

Related to the effects of numerous small ponds across the landscape, the FCS structures likely have a tremendous effect on flows in the Lampasas River. Similar structures have exhibited substantial reductions in peak flood discharge (Tortorelli, 1997, Van Liew *et al.*, 2003). Indeed, these structures were designed for this very purpose. In addition to reducing peak flows, they have also been suggested as sites of increased infiltration due to high retention times, potentially boosting baseflows. However, in other settings, the presence of an FCS has been linked to small decreases in streamflow throughout the year, including during dry periods, due to evaporation of retained runoff (Van Liew *et al.*, 2003). Ultimately, the hydrological effect of specific FCS sites is dependent upon local characteristics of precipitation, temperature, soil, and geology. In nearly all cases, these dams influence flows on both the upper and lower ends of the hydrograph, though on a much larger scale than small farm ponds. In general, the hydrological impact of dams on streamflows can be many times greater than likely effects from climate change (Graf, 1999). Ponds and lakes surely exert some influence on the Lampasas River and its tributaries.

Chin *et al.* (2008) identified a positive correlation between small dams and population centers in Texas. Though this excludes the small ponds constructed on agricultural lands, it does include the FCS reservoirs, which were constructed to prevent downstream flooding of population centers and associated infrastructure. The very presence of these structures in proximity to human population centers raises the possibility that land management practices and water use patterns may be somewhat different than those farther removed from urban centers, possibly affecting the watershed processes in these areas.

Trends in streamflow are correlated with groundwater levels in most landscapes (Esralew & Lewis, 2010). Changes in groundwater usage can greatly affect water resources, even at large scales. Agricultural development and urban expansion to accommodate regional population growth have increased irrigation demand with major negative effects on aquifers in many regions of the world (Gleick, 2010). However, improved crop production techniques and land management practices can also improve infiltration and groundwater recharge to a moderate extent. A large-scale study streamflow trends in Oklahoma indicated an upward trend in each baseflow and total streamflow at most gaging stations examined (Esralew & Lewis, 2010). Changes were closely linked to geography within the state and related to changing local water use and land management practices over time, including local irrigation demands. Most of the state experienced increases in groundwater levels stemming from reduced groundwater withdrawals over the last few decades. Irrigated agriculture is almost nonexistent in this portion of the Lampasas Cut Plain, and large reductions in cropland area would not result in lower groundwater withdrawals for this purpose.

Where groundwater levels and baseflow have risen concurrently, one possible explanation for increases in the long-term recharge, driven by elevated precipitation, occurring much more gradually than responses in streamflow (Esralew & Lewis, 2010).

However, this assumes groundwater-surface water interactions are sufficient to lead to increases over time. As seen in Figure 5.2, streamflow at the downstream Youngsport gaging station was frequently lower than that observed at the Kempner gaging station, despite having a much greater catchment size. This behavior of a losing stream segment suggests a significant connection between surface flows and underlying groundwater over the course of the lower Lampasas River, signaling another complicating factor in assessing hydrologic trends in the basin. Wilcox and Huang (2010) noted that, due to groundwater connections and associated outcrops, the effective catchment for some watersheds may be larger and much different than delineations on the surface. In this vein, streamflows at Kempner, which are composed largely of runoff from springs near Lampasas, may reflect landscape conditions over a much larger area than that which we examined here. The Marble Falls aquifer that supplies springflow to Sulphur Creek, underlies a large area, most of which is outside the Lampasas River basin. As a result, streamflows recorded in this study likely reflect hydrological conditions over an area many times the size of the watersheds we examined.

Furthermore, there is a tremendous amount of interannual variability in both precipitation and streamflow in the Lampasas Cut Plain, even in the midst of cyclical trends over broader periods. The southern Great Plains is characterized by boom-or-bust cycles of drought and high precipitation and runoff, largely tied to El Niño-Southern Oscillation anomalies (Dunbar *et al.*, 2010). An examination of Lampasas River baseflow characteristics conducted following a prolonged dry period found that nearly all tributaries to the main stem cease to flow during drought. Only Sulphur Creek, which drains the Lampasas County watersheds examined here, and Salado Creek downstream of the former Youngsport gaging station, exhibited significant flow under dry conditions (Mills & Rawson, 1965). Given that the unique geology that supplies springs in each of these locations is quite different than the majority of the immediate area, the hydrology of much of the remaining watershed appears to be dominated by overland runoff processes. One early study of rainfall and runoff mechanisms in the Lampasas River basin suggested that annual streamflow as a proportion of total precipitation generally ranged between 4-7% (Tanner *et al.*, 1937). This figure is very close to our estimated long-term average of 8%. Especially when compared with much higher rates from the Edwards Plateau, this figure suggests a great deal of internal storage within the landscape and much higher rates of loss to evapotranspiration in the Lampasas Cut Plain.

Pulling together all of these considerations of changes in woody plant cover, construction of small impoundments, groundwater inputs from outside the basin, and climatic variability, it is very difficult to make any definitive conclusions about hydrological processes in the Lampasas River and the surrounding landscape. It is probable that some of these factors work in concert with and oppose other drivers of local hydrological response. We cannot obtain streamflow data prior to the instrumental record and therefore are not able to make reliable inferences of watershed function in the absence of small ponds and prior to current land cover patterns. However, the long-term stability of precipitation and streamflows in this area are remarkable and suggest plant cover and composition may be a very minor determinant of local water yields.

Most studies investigating the effect of dams on sediment and water flows have focused on large structures, and many questions remain with respect to medium- and small-sized structures. Fewer studies still have examined these impoundments in Texas, though 97% of the dams in the state impound less than 10 million m^3 (Chin *et al.*, 2008). In addition to providing information on historical conditions and processes in their respective watersheds, FCS reservoirs serve to retain sediment that would otherwise be transported downstream to larger drinking water reservoirs, preserving capacity and prolonging their lifespan (R. J. Brandes Company, 2011). These dueling effects of reductions in streamflow and sediment transport prevent both flows from reaching downstream sites. The cumulative effect in some locations has been shown to maintain the storage capacity of downstream reservoirs yet reduce firm annual water yield significantly, with greatest effects in more arid climates.

While this study primarily focuses on the dynamics of rangelands themselves, it provides an important look into the processes that directly affect downstream water resources, which themselves drive management decisions on local agricultural lands. There are a number of inherent feedbacks and cascading effects in these systems. For instance, overgrazing by livestock may increase erosion and lead to significant sedimentation of impoundments immediately downstream. These impoundments often serve as a water supply for the livestock, which is diminished as storage capacity is reduced through deposition. As a result, an unsustainable concentration of livestock has the potential to reduce further the carrying capacity of the landscape both through loss of soil and filling of water resources. While it does not appear that increases in woody plant cover result in changes in water yield or sedimentation in this landscape, there does appear to be potential for increases in sediment yield associated with drought and land use intensification that can affect downstream water storage. Streamflow and sediment responses to changing landscape conditions involve the integration of many and complex processes across a watershed. Though the direct causes of shifts in hydrological and sediment dynamics may be difficult to draw out, an understanding of how rangelands work is critical. Identifying long-term trends in landscape characteristics and hydrologic response plays a vital role in water resource planning and management and allocating water to different uses at all scales.

CHAPTER VI

CONCLUSIONS

The approach employed here, including both large-scale application of remote sensing imagery classification and targeted analyses of local hydrological and sediment records, represents a very high level of investigation into the history and processes at work in rangelands in central Texas. While a number of studies have addressed the spatial considerations of woody plant encroachment and others have identified trends in hydrological response and sediment yield to changing landscape conditions, there has been almost no documentation of the link between these variables at the watershed scale, much less incorporating measures of sociological influence on land cover and land use change. We have developed an innovative and thorough history of 230 km^2 of the Lampasas Cut Plain in Lampasas, Mills, and Burnet Counties.

Despite widespread anecdotal accounts of uniform encroachment of woody plants (primarily Ashe juniper), the story is much more complex in reality. Indeed, cover by shrubs and trees was very high locally in the 1930s and 1940s, even more extensive than present conditions in some watersheds. Additionally, with the exception of a peri-urban watershed near the Lampasas city limits, every watershed experienced a significant decline of woody plant cover from the initial time period. However, while watersheds in Mills County remained at low levels of woody plant cover, those in Lampasas County rebounded to shrub levels near or somewhat above those in 1940.

These dramatic and different changes of woody plant extent were observed in the context of broader landscape change that reflected similar changes across much of the southern Great Plains. In the counties and watersheds examined here, there has been a dramatic conversion of land use tendencies, away from cropland production to rangeland use. Nearly 80% of the cropland area in this region was lost between the 1930s and 2012. At the same time, there has been a significant increase in the number of constructed small farm ponds across the landscape in these watersheds, correlated to the shift away from crop production to rangeland and wildlife uses. The number of ponds has increased almost fourfold since the 1930s and 1940s, with Mills County attaining greatest densities by 2012. Many ponds constructed in each county had undergone maintenance activities included enlarging and excavation, with some receiving attention on a recurring basis.

While these trends are important by their very nature, an understanding of their drivers is critical to making policy recommendations for changing landowner priorities and management objectives. Our study provides one of the first connections between woody plant cover trends and those of human factors such as population. In each case, be it steadily decreasing shrub cover in Mills County, steadily increasing shrub cover on the edges of Lampasas, or the declining shrub cover followed by a rebound of similar magnitude in rural Lampasas County, human populations followed an almost identical pattern to that of woody plants, both in time and in scale of change. This establishes management and land ownership trends as significant factors in landscape change in central Texas beyond the purely ecological factors typically described as being of major importance.

Finally, though these landscapes have undergone a great deal of nonlinear change in terms of land cover and land use, as well as related human variables, the watersheds examined here appear to be remarkably resilient to change with regard to their dynamic processes. Both hydrologically and in terms of sediment delivery, little has changed in this area. Since the 1920s, streamflows have remained almost identical across the area. Sediment delivery generally displays a similar lack of change in most watersheds, though the timeframe of observation is constrained to approximately 1960 in most cases and as recently as the late 1970s in others. Nonetheless, the surprising lack of landscape response at the watershed scale to plant cover and management changes is critical to implementing conservation measures to maximize ecosystem services delivered by these areas. However, a great deal of continuous monitoring is necessary to stay ahead of any threshold response that may trigger dramatic changes in watershed function in cases of significant alterations in local land use and land cover.

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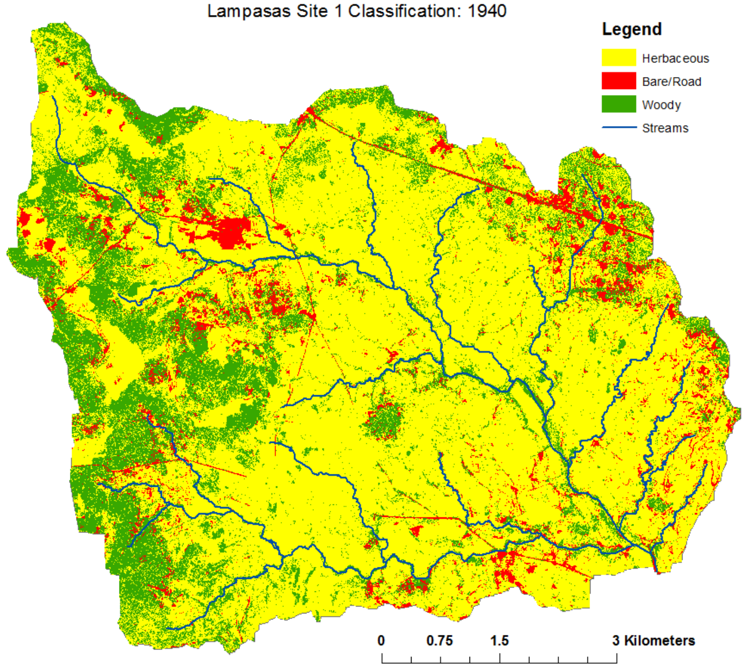
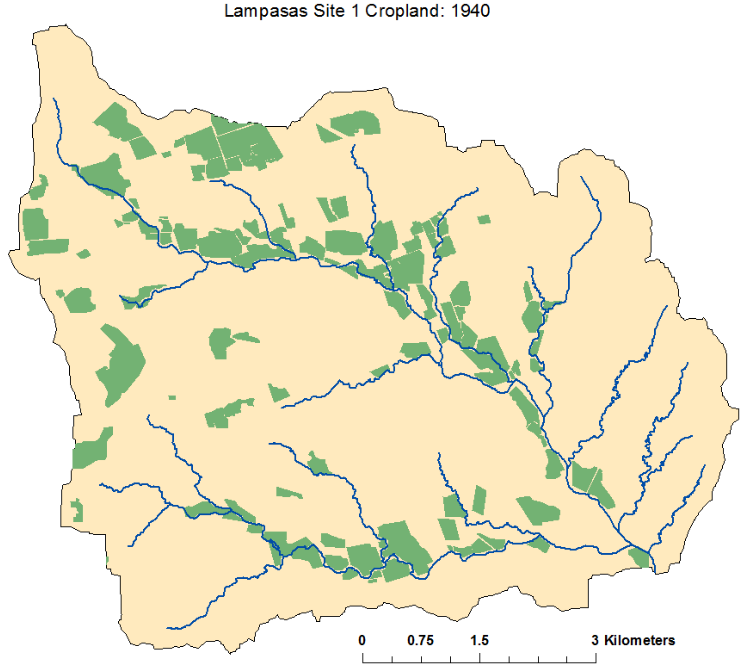
APPENDIX A

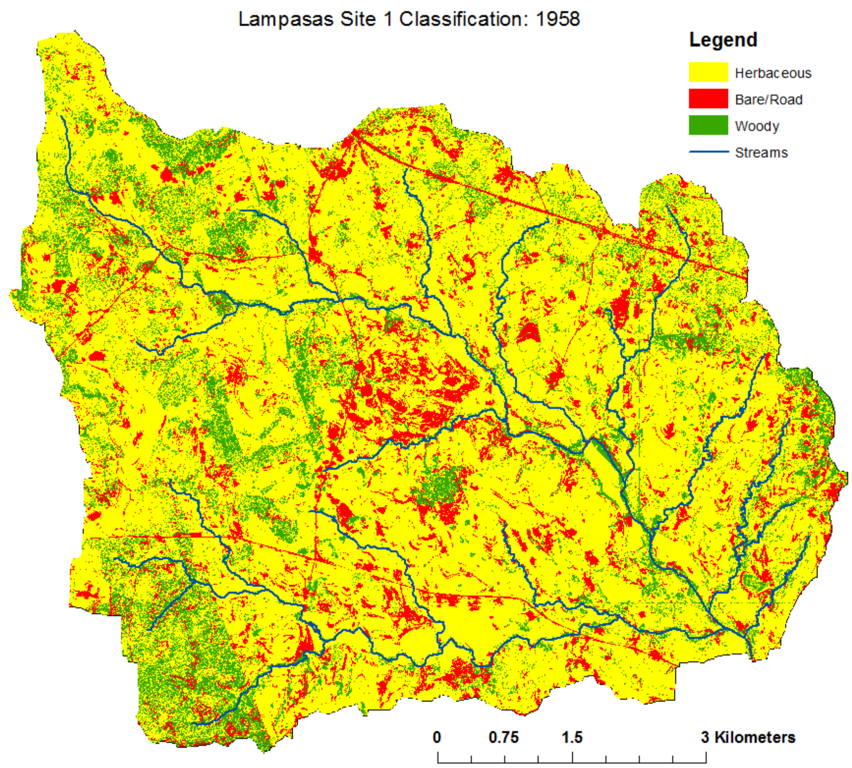
WATERSHED LAND COVER AND CROPLAND EXTENT BY YEAR

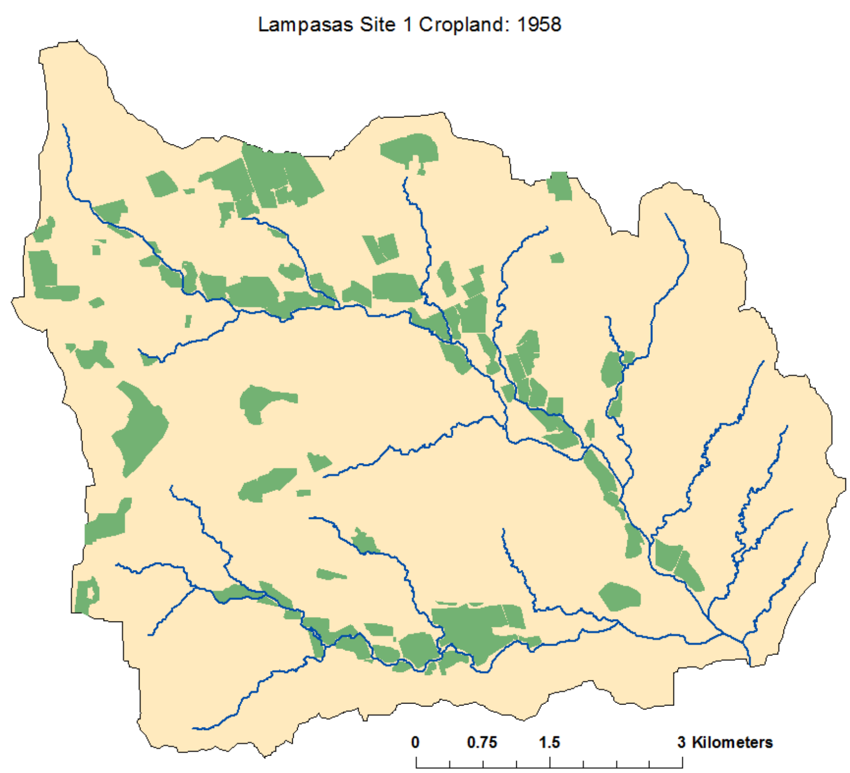
Lampasas Site 1 Classification: 1940

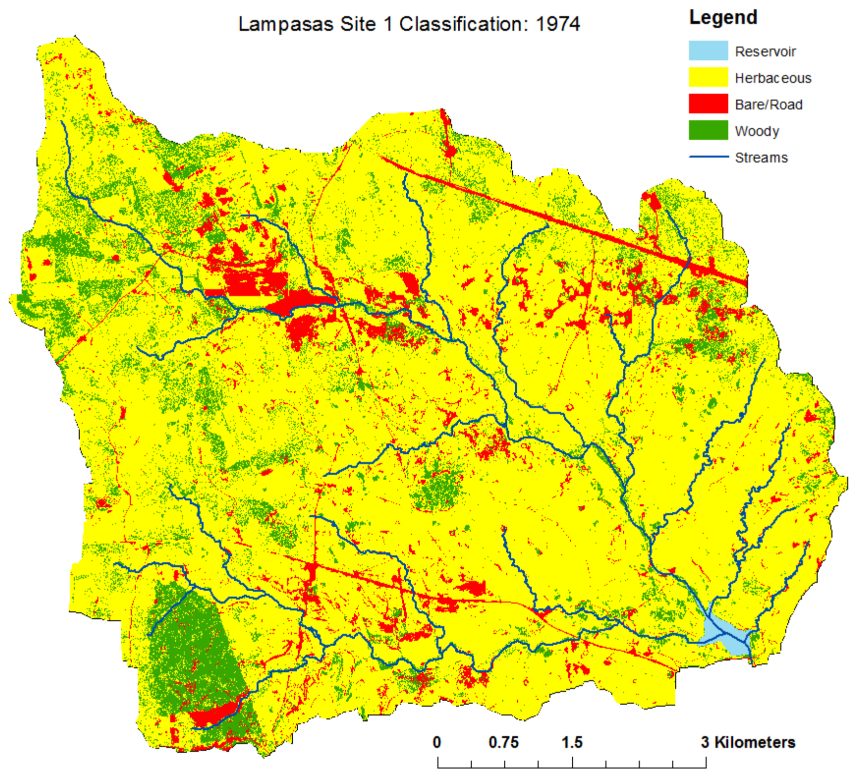


Lampasas Site 1 Cropland: 1940

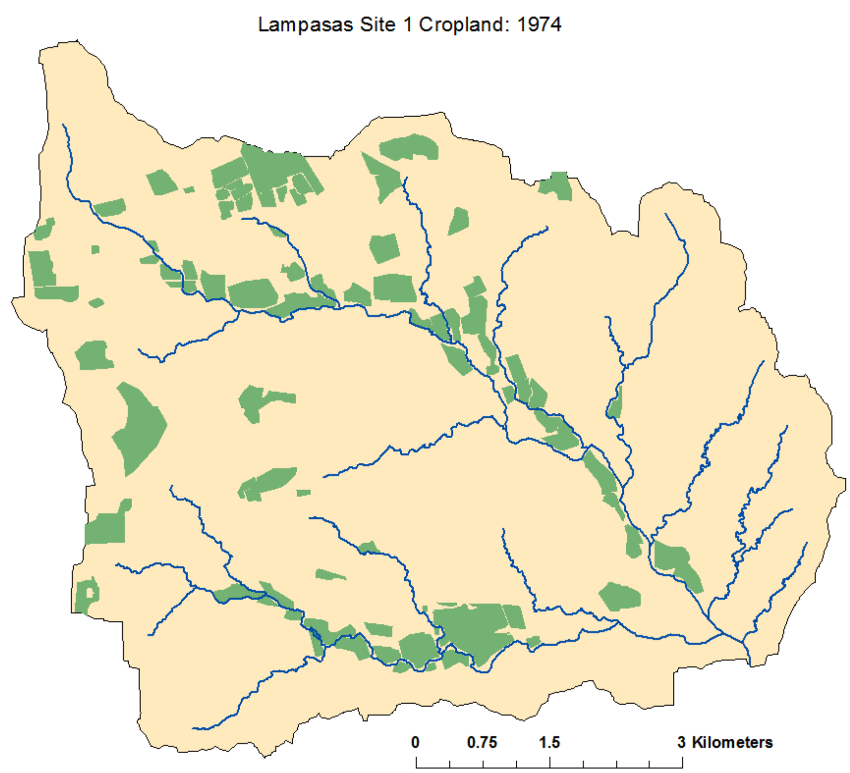



Lampasas Site 1 Classification: 1958
Legend
Herbaceous
Bare/Road
Woody
Streams
0
0.75
1.5
3 Kilometers

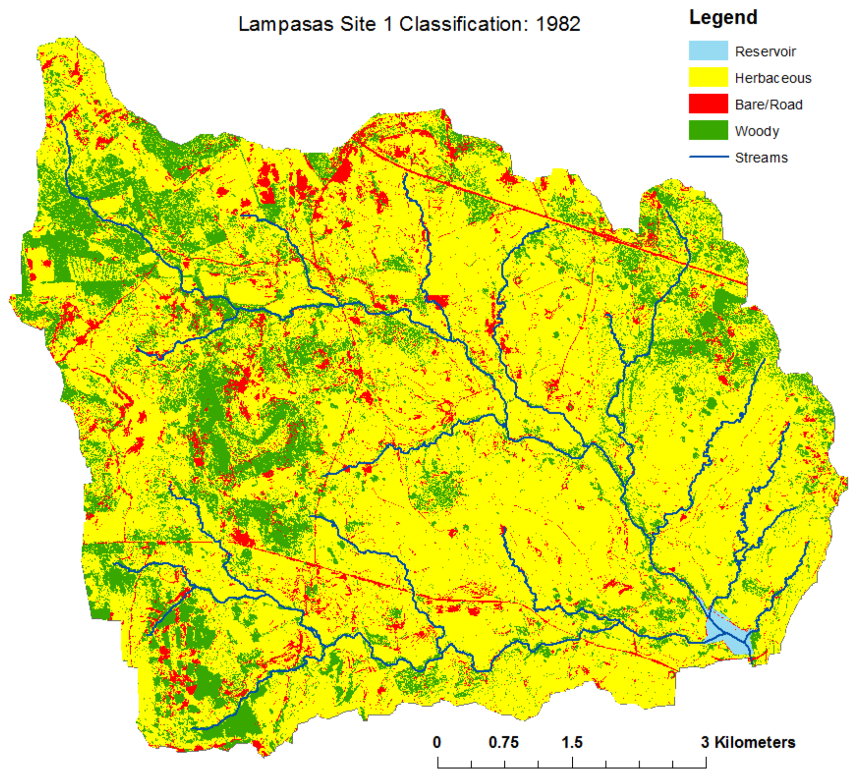

Lampasas Site 1 Cropland: 1958
0
0.75
1.5
3 Kilometers



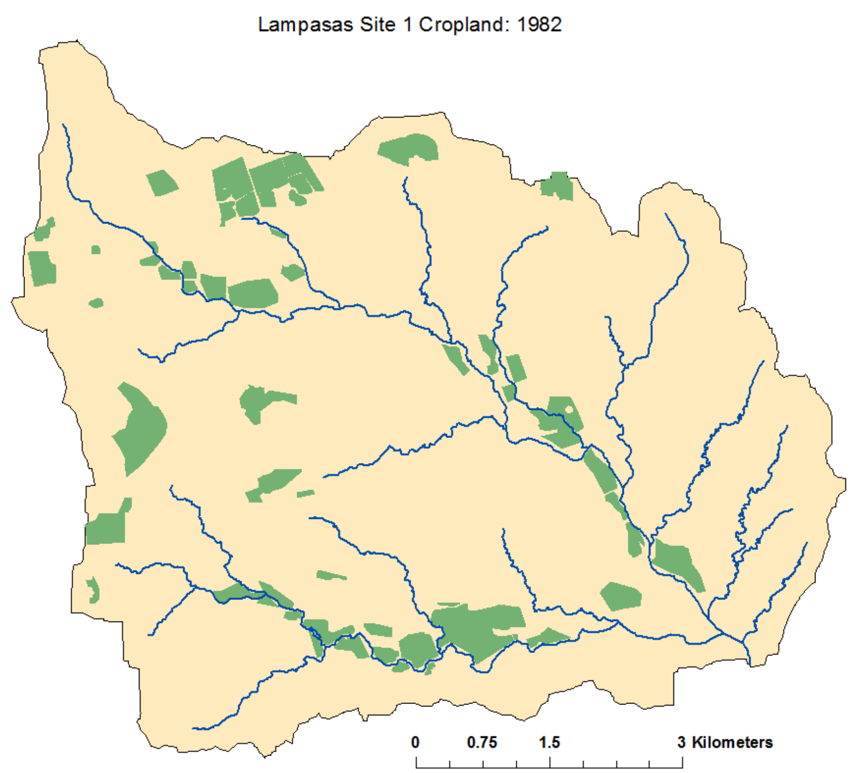
Lampasas Site 1 Classification: 1974
Legend
Reservoir
Herbaceous
Bare/Road
Woody
Streams
0
0.75
1.5
3 Kilometers



Lampasas Site 1 Cropland: 1974
0
0.75
1.5
3 Kilometers

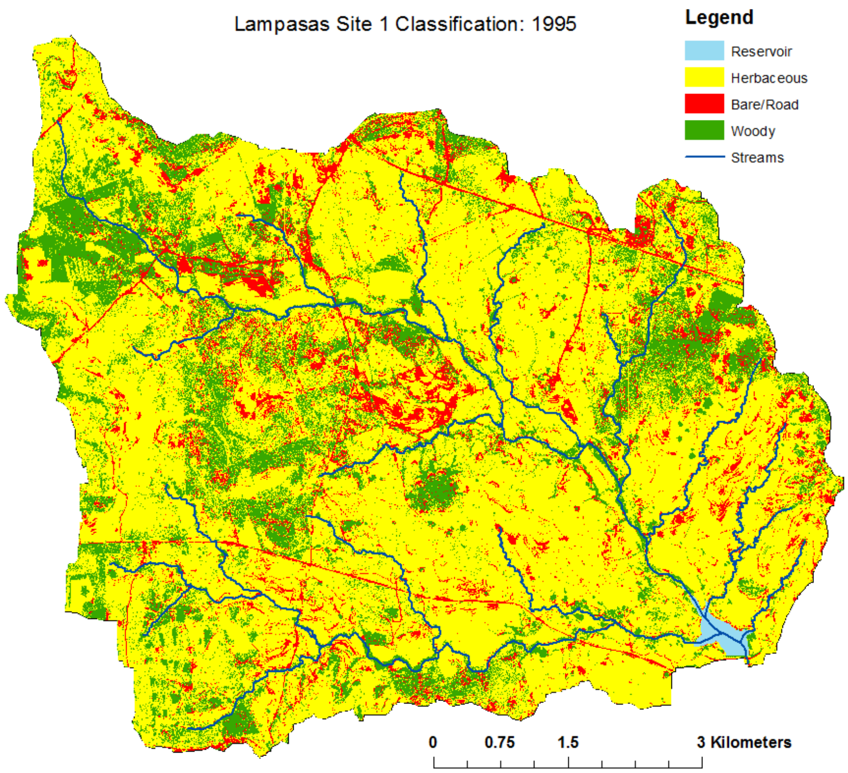


Lampasas Site 1 Classification: 1982
Legend
Reservoir
Herbaceous
Bare/Road
Woody
Streams
0
0.75
1.5
3 Kilometers

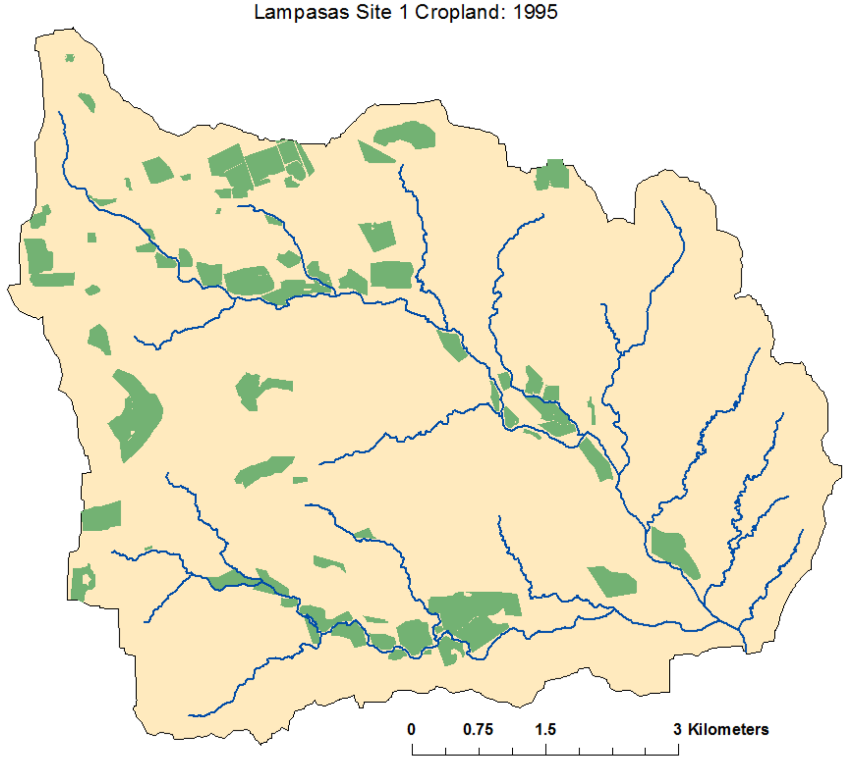


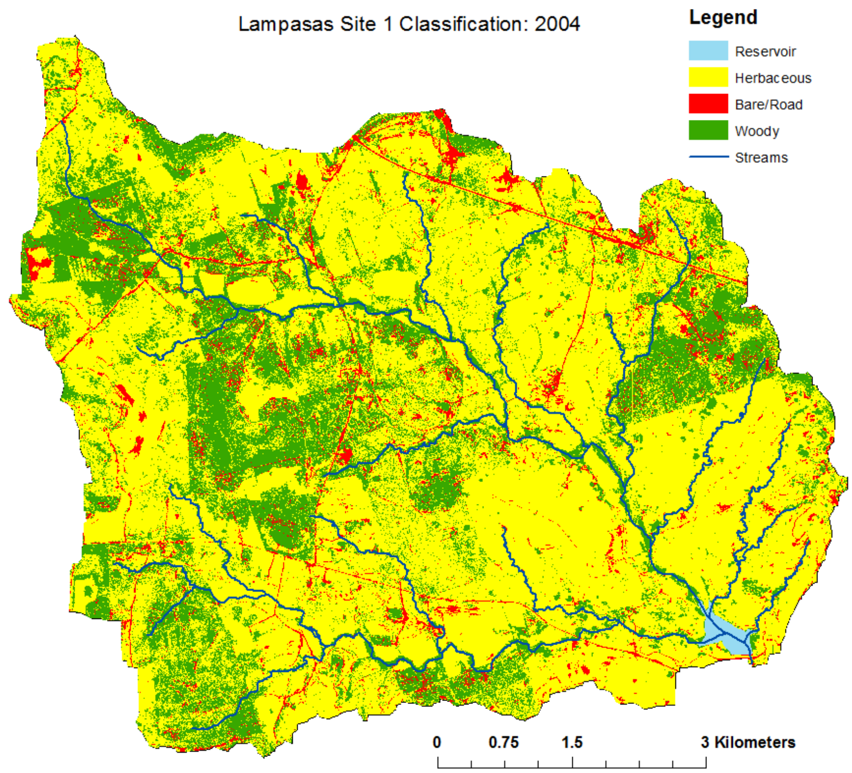
Lampasas Site 1 Cropland: 1982
0
0.75
1.5
3 Kilometers

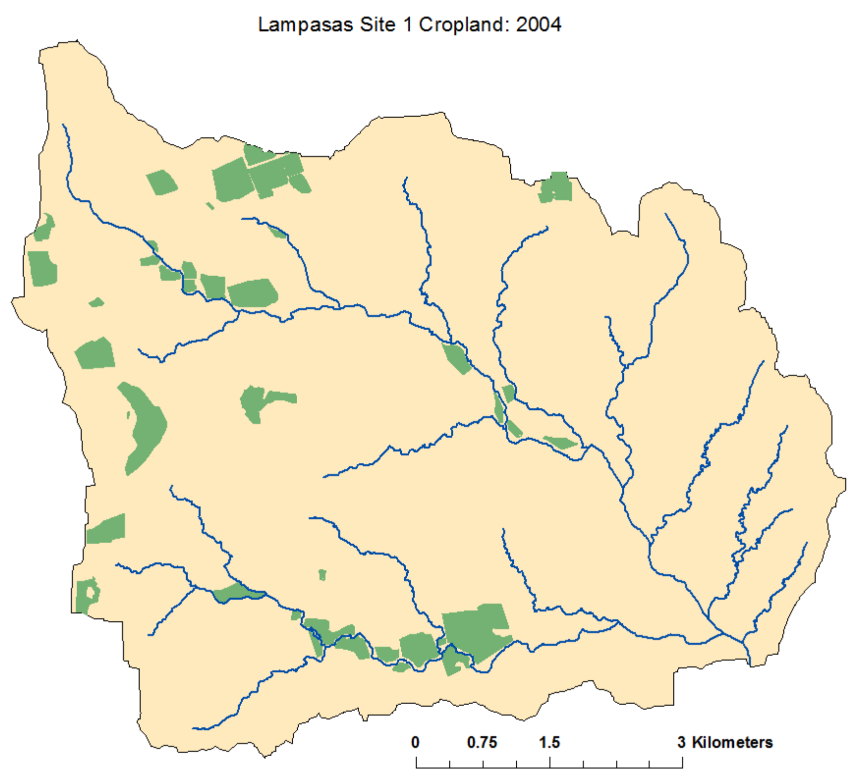
Lampasas Site 1 Classification: 1995

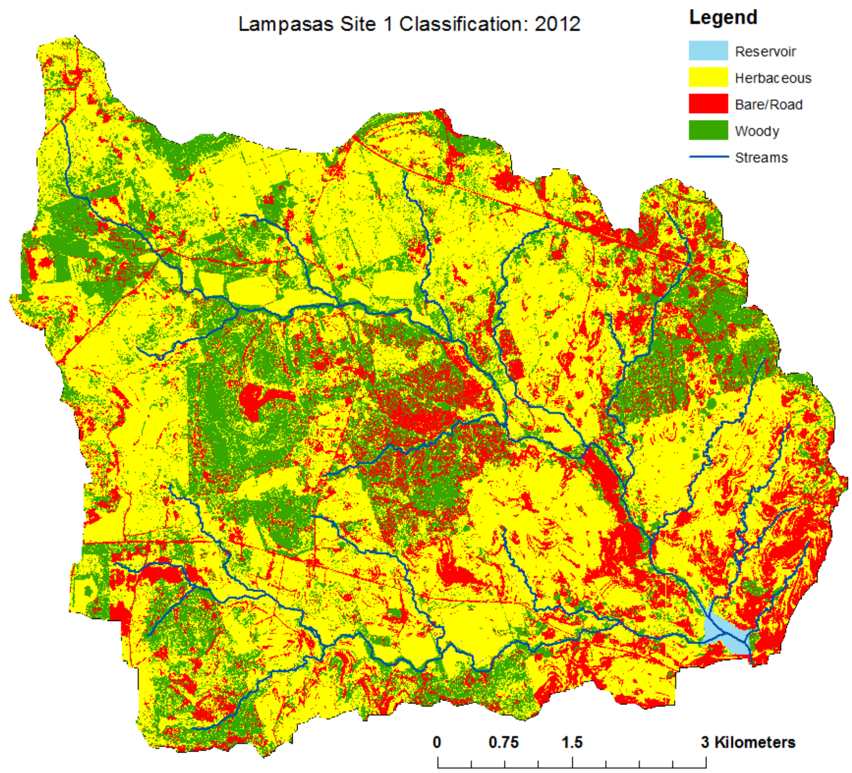


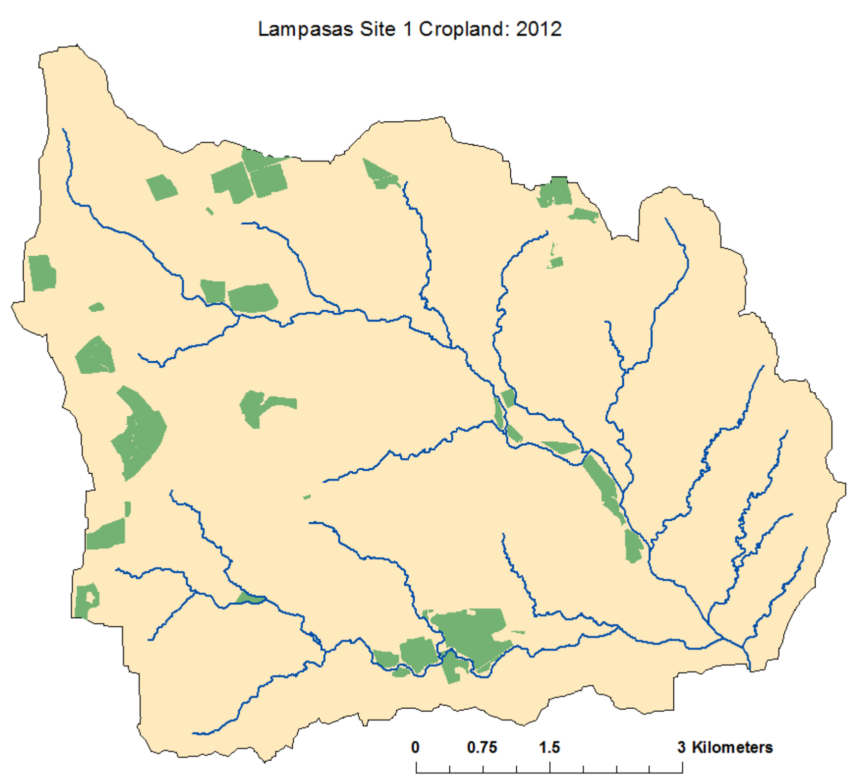
Lampasas Site 1 Cropland: 1995



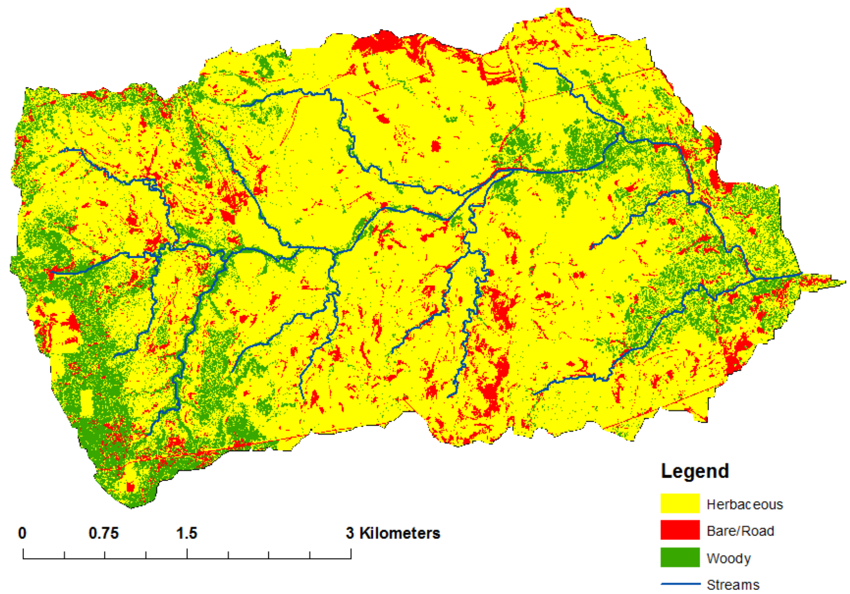

Lampasas Site 1 Classification: 2004
Legend
Reservoir
Herbaceous
Bare/Road
Woody
Streams
0
0.75
1.5
3 Kilometers


Lampasas Site 1 Cropland: 2004
0
0.75
1.5
3 Kilometers

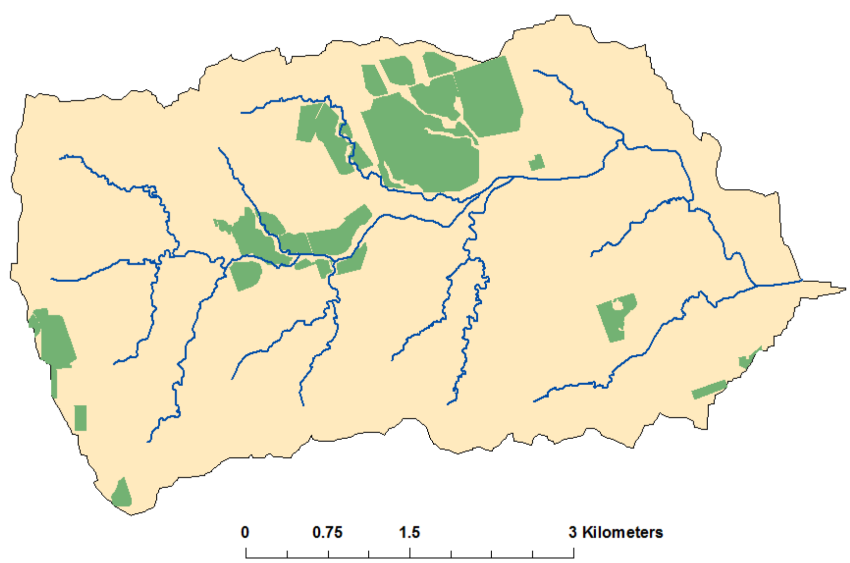

Lampasas Site 1 Classification: 2012
Legend
Reservoir
Herbaceous
Bare/Road
Woody
Streams
0
0.75
1.5
3 Kilometers


Lampasas Site 1 Cropland: 2012
0
0.75
1.5
3 Kilometers

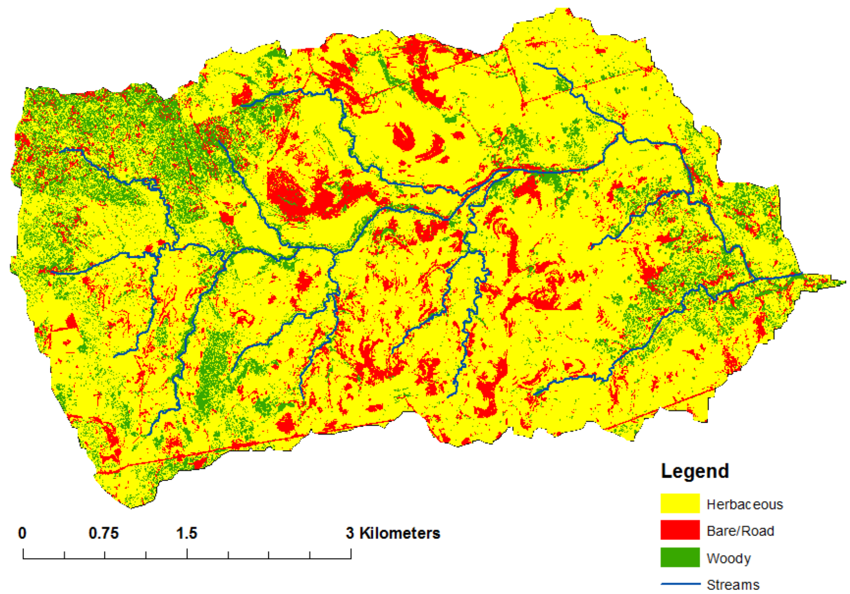
Lampasas Site 2 Classification: 1940



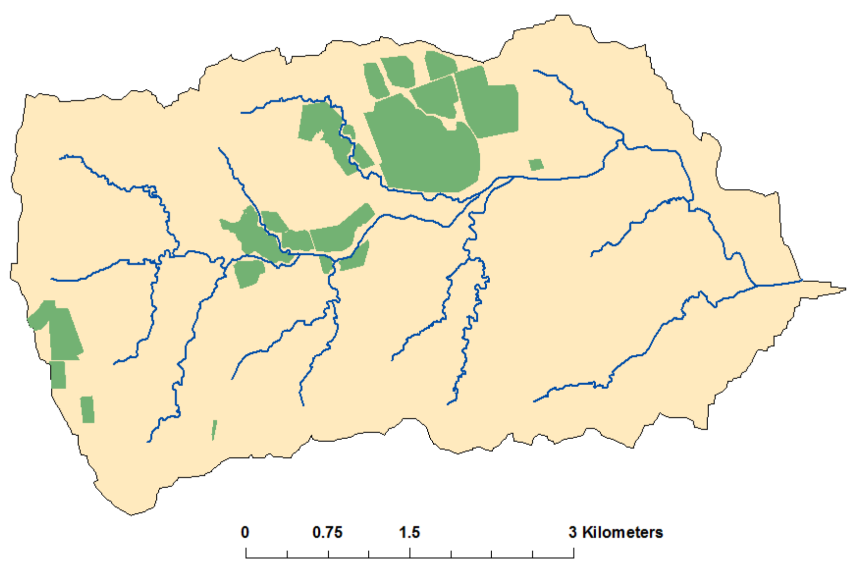
Lampasas Site 2 Cropland: 1940



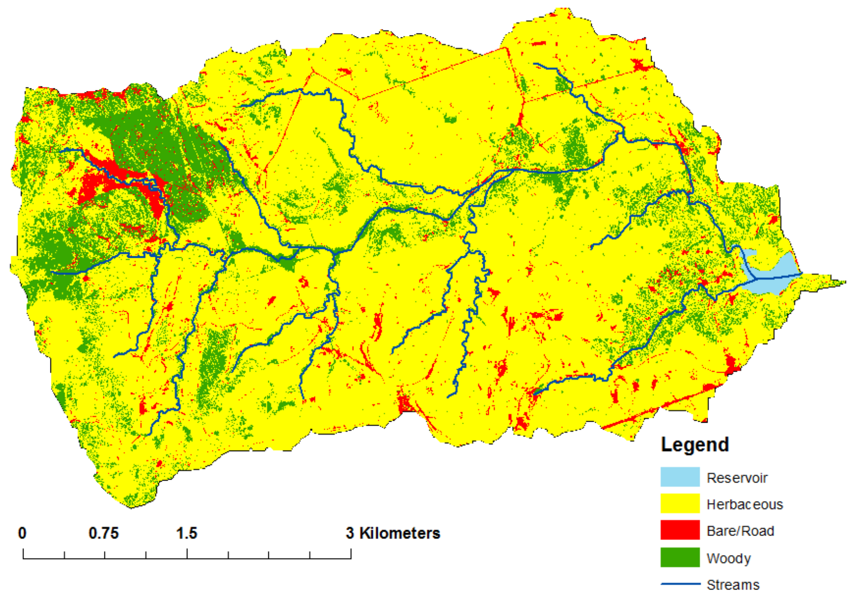
Lampasas Site 2 Classification: 1958



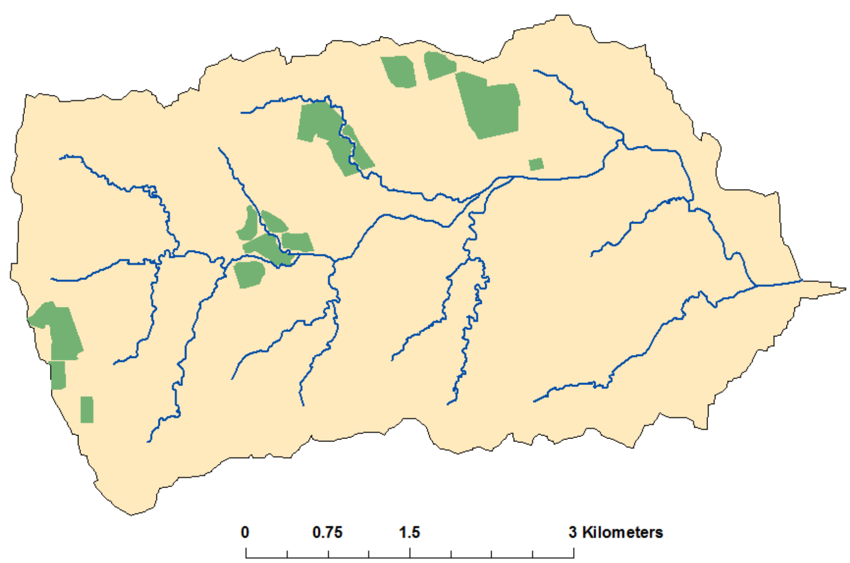
Lampasas Site 2 Cropland: 1958



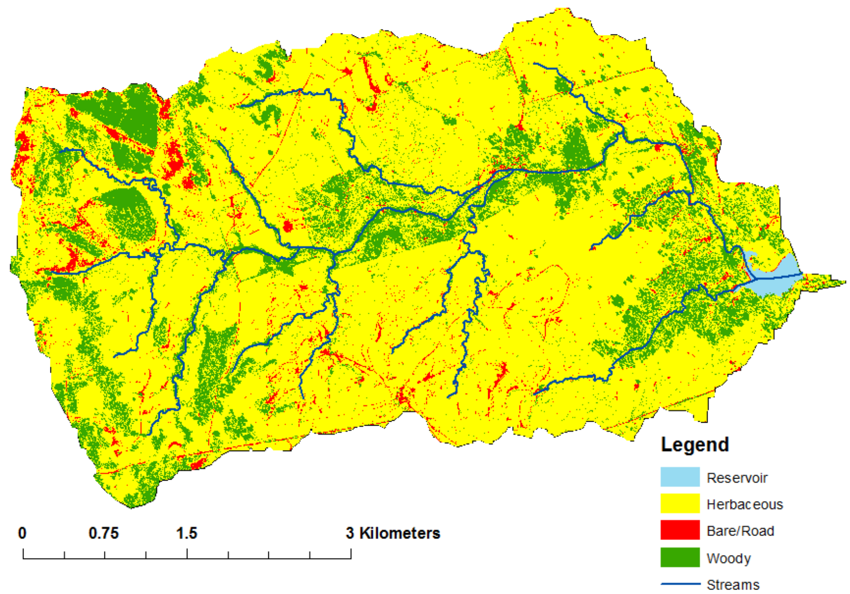
Lampasas Site 2 Classification: 1974



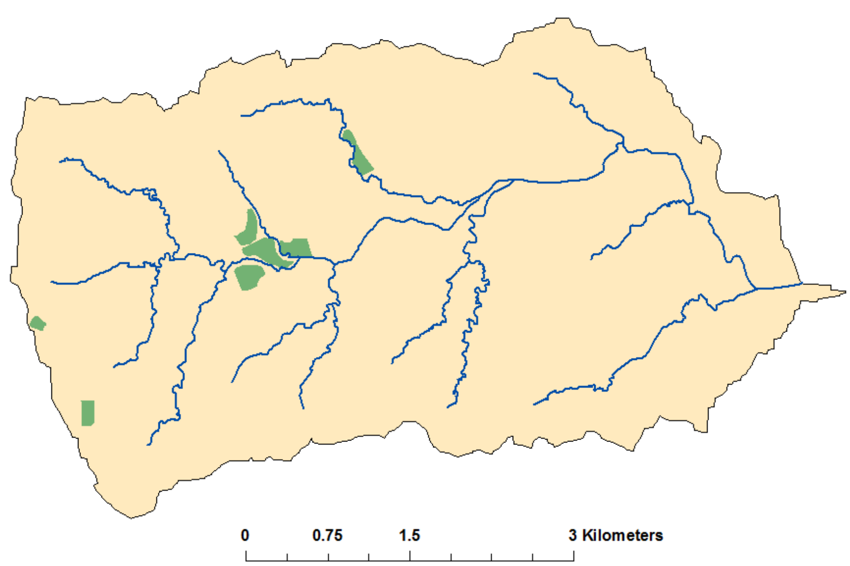
Lampasas Site 2 Cropland: 1974



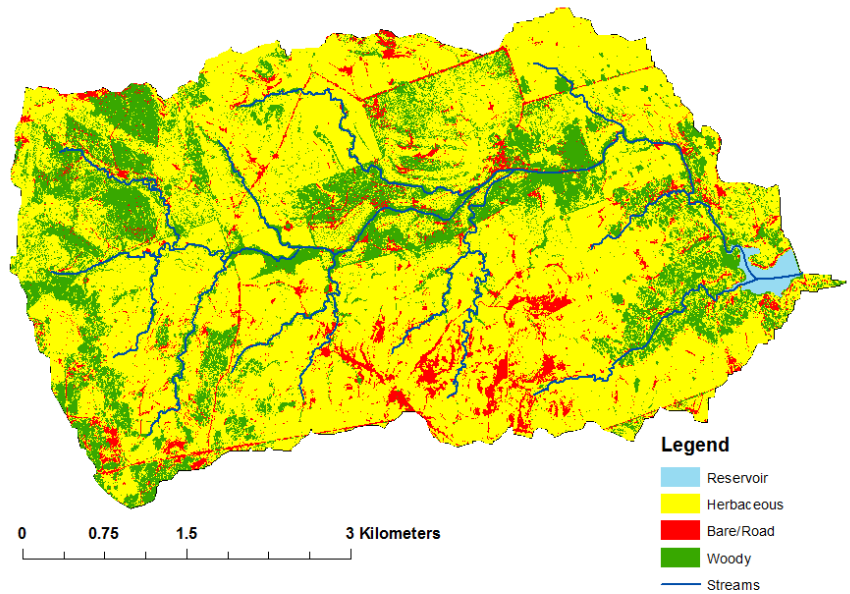
Lampasas Site 2 Classification: 1982



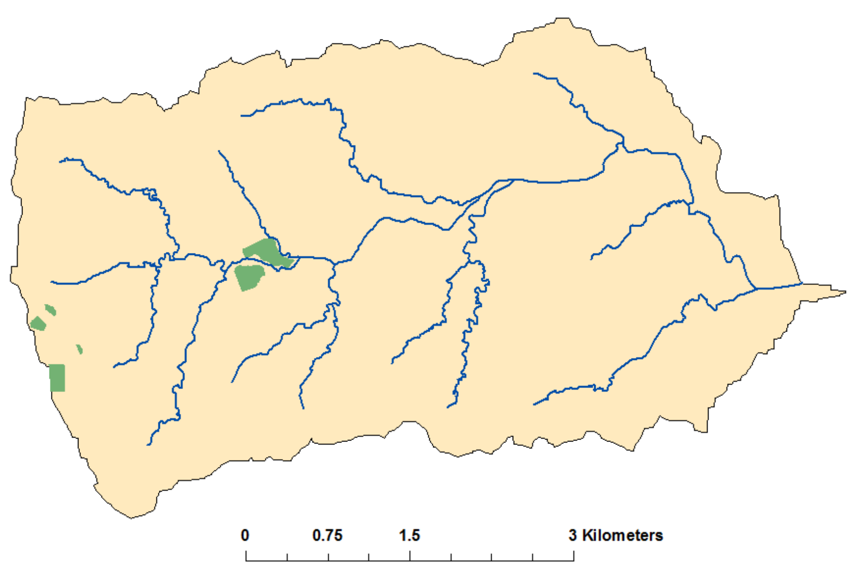
Lampasas Site 2 Cropland: 1982



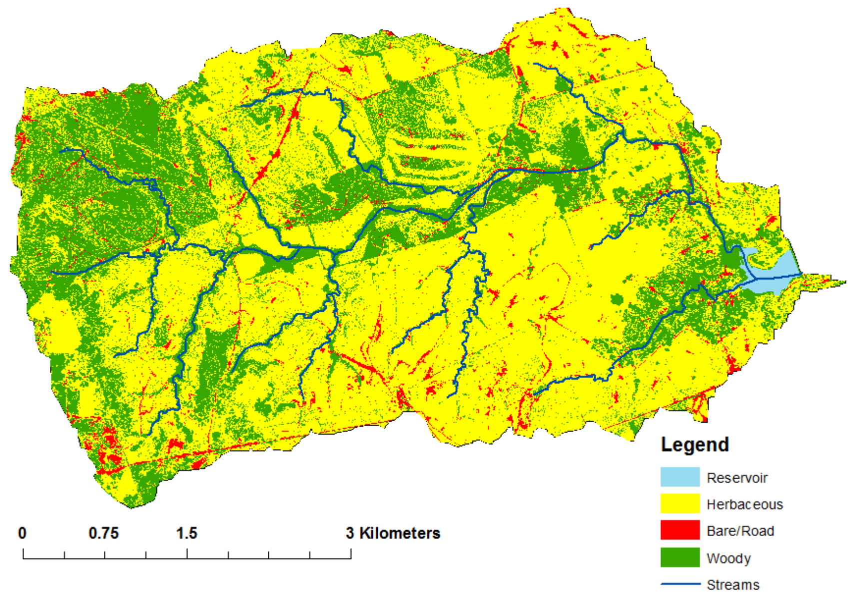
Lampasas Site 2 Classification: 1995



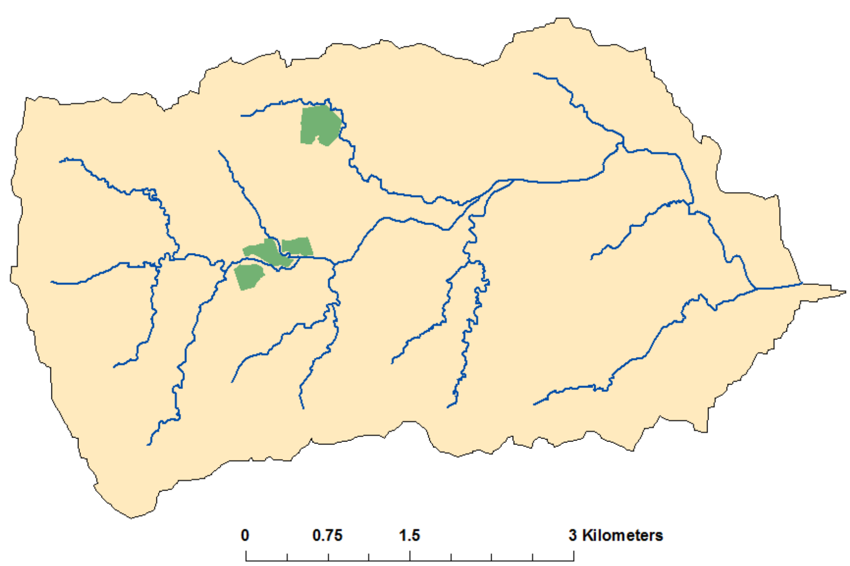
Lampasas Site 2 Cropland: 1995



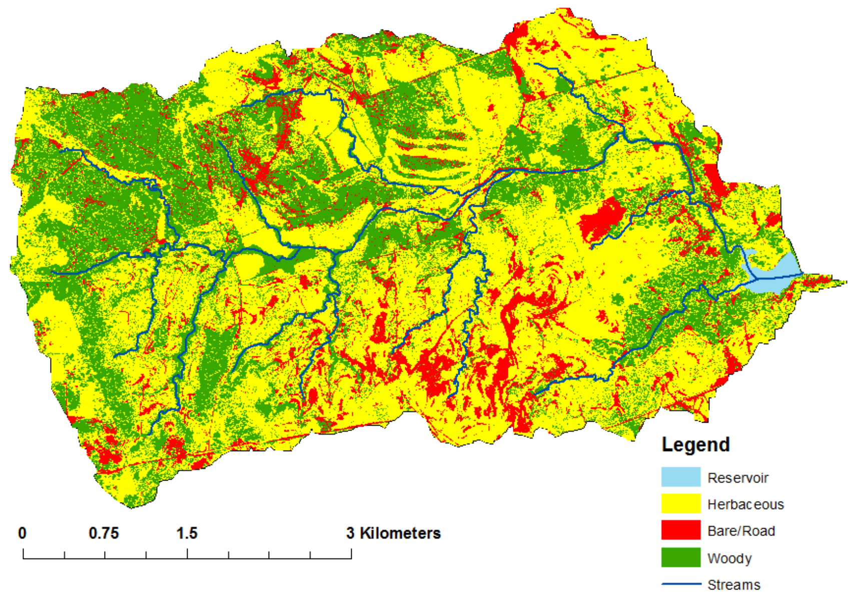
Lampasas Site 2 Classification: 2004



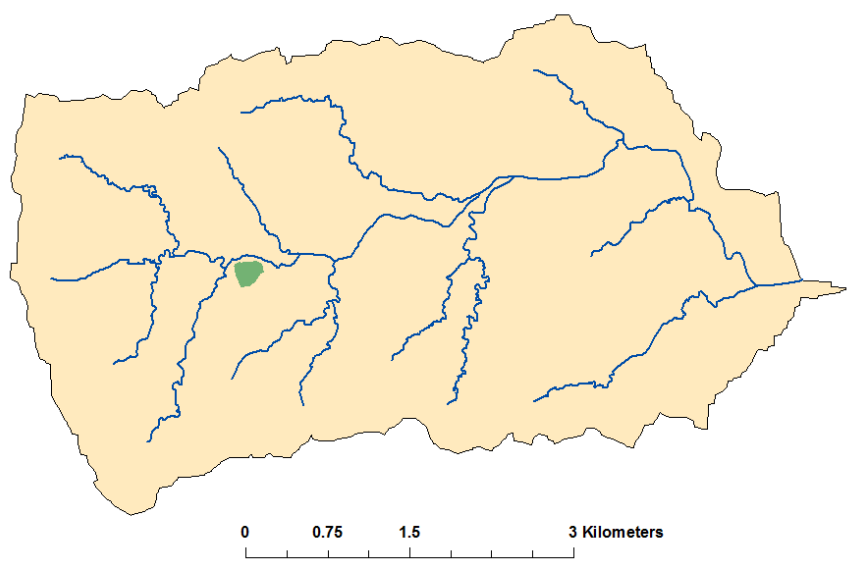
Lampasas Site 2 Cropland: 2004



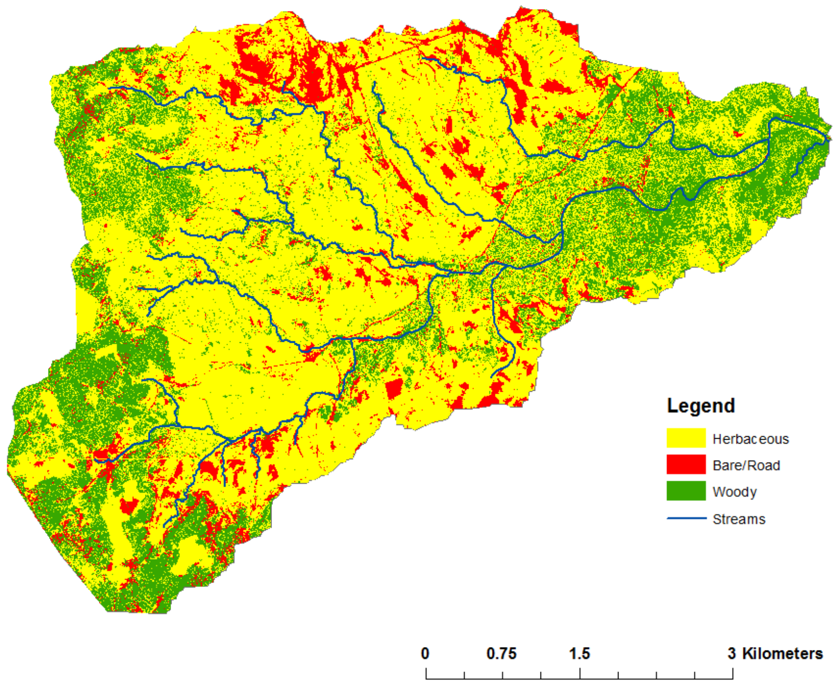
Lampasas Site 2 Classification: 2012



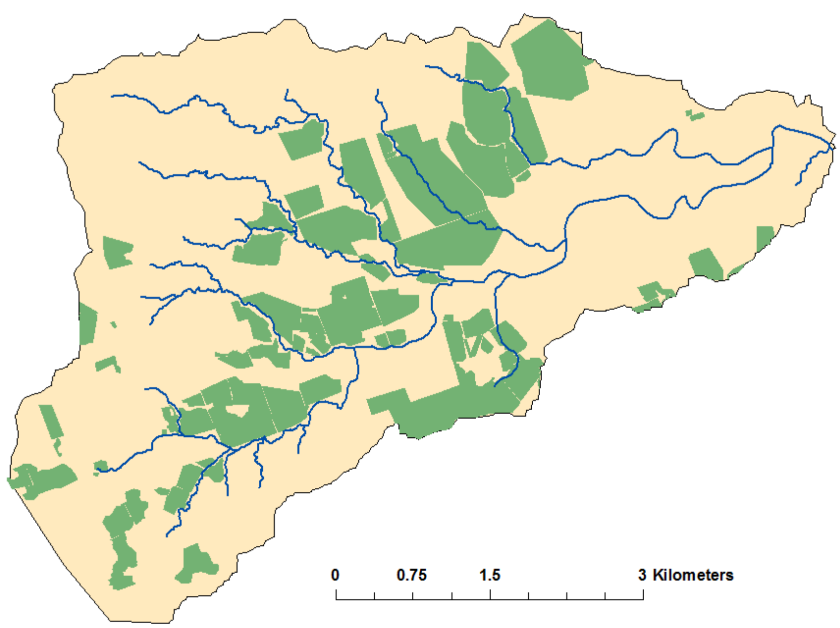
Lampasas Site 2 Cropland: 2012



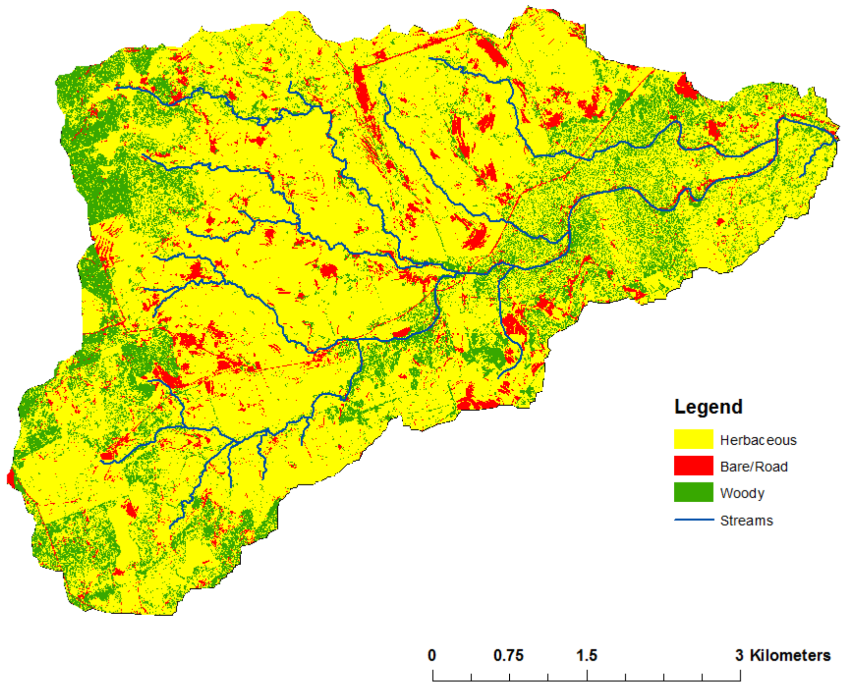
Lampasas Site 3 Classification: 1940



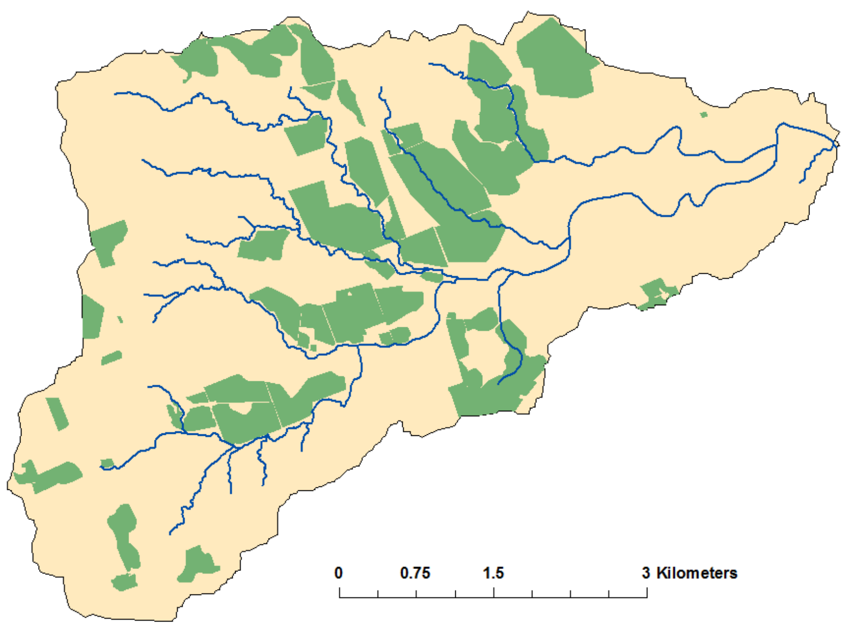
Lampasas Site 3 Cropland: 1940



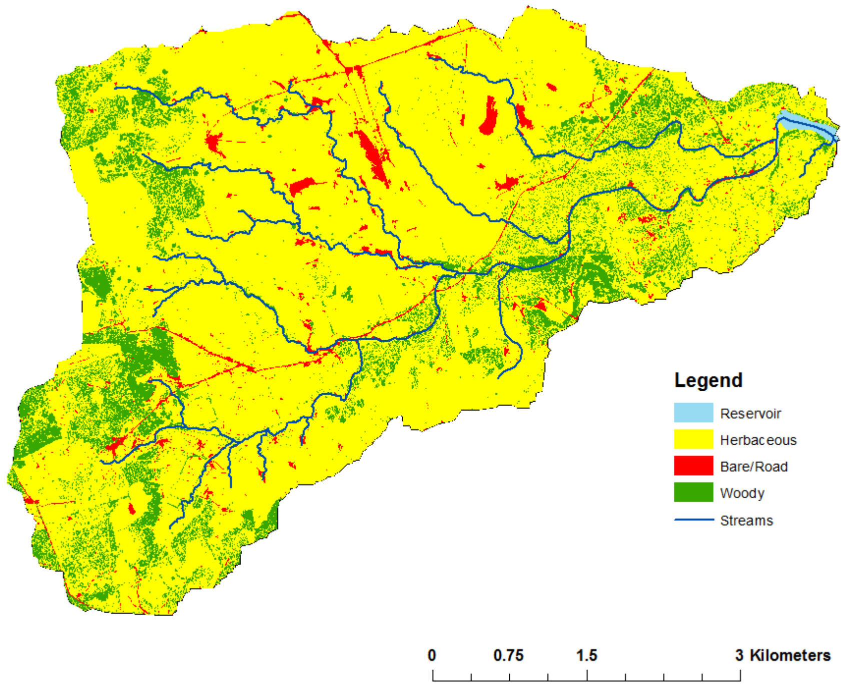
Lampasas Site 3 Classification: 1958



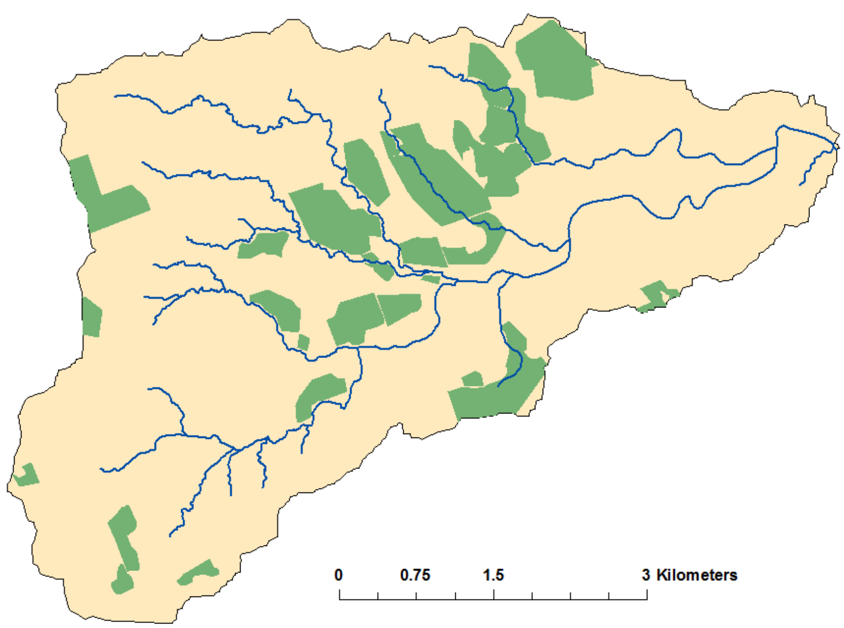
Lampasas Site 3 Cropland: 1958



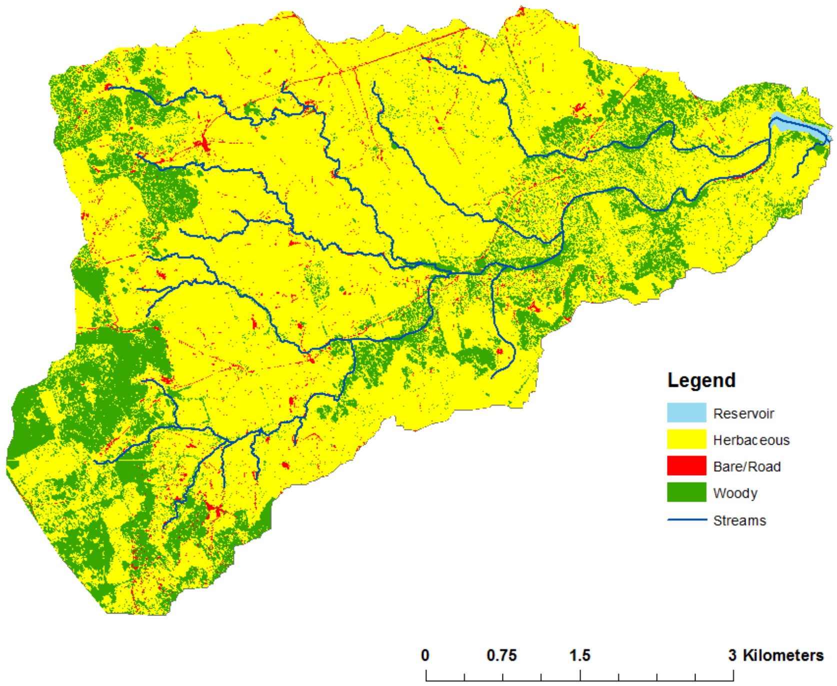
Lampasas Site 3 Classification: 1974



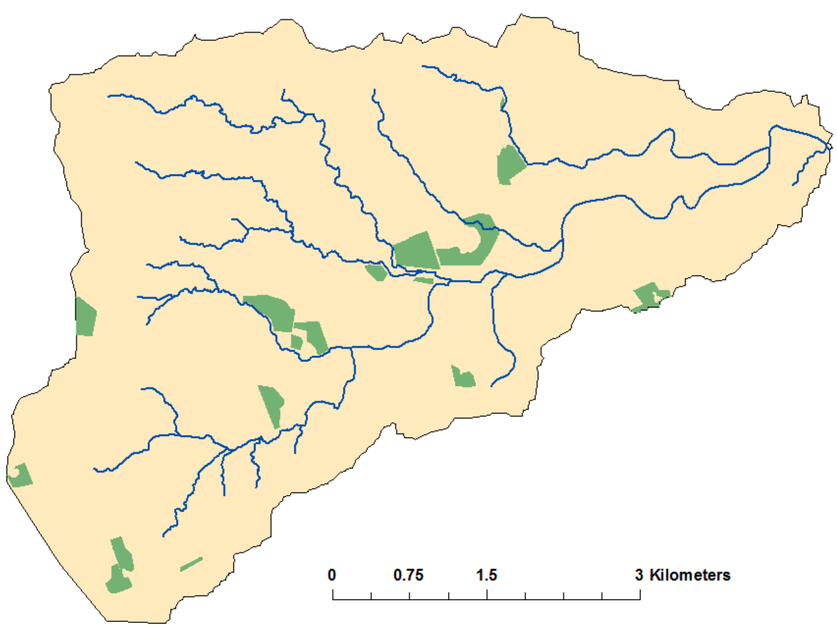
Lampasas Site 3 Cropland: 1974



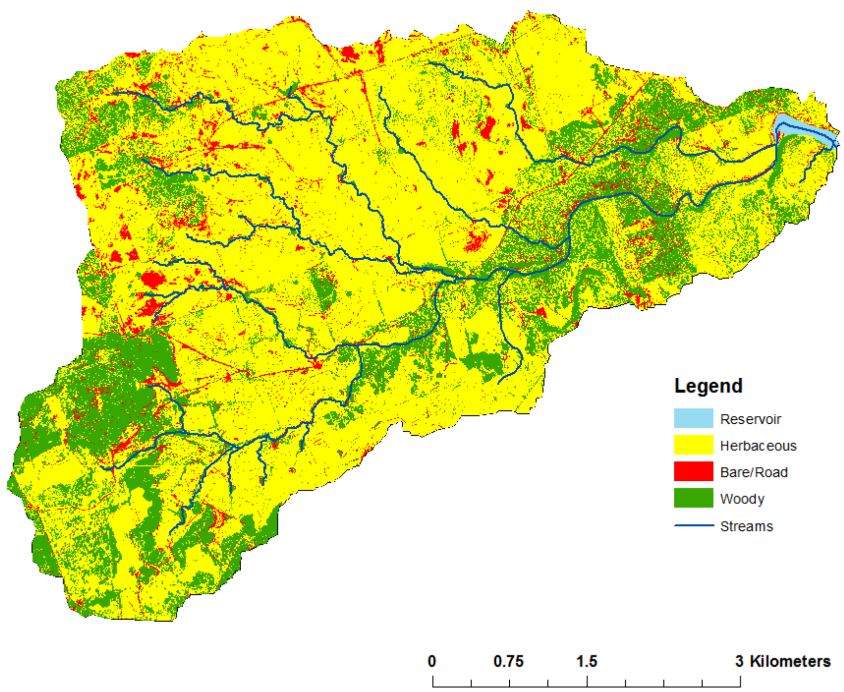
Lampasas Site 3 Classification: 1982



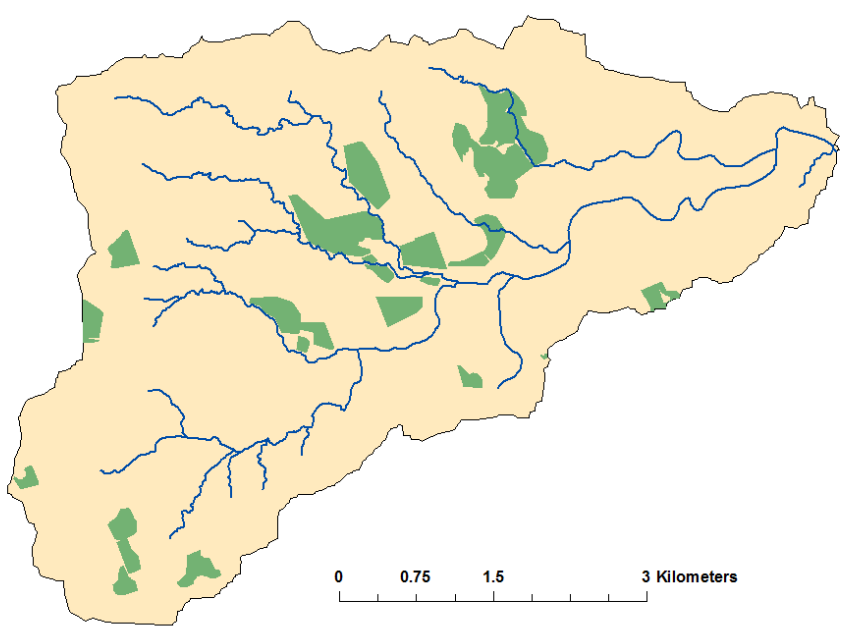
Lampasas Site 3 Cropland: 1982



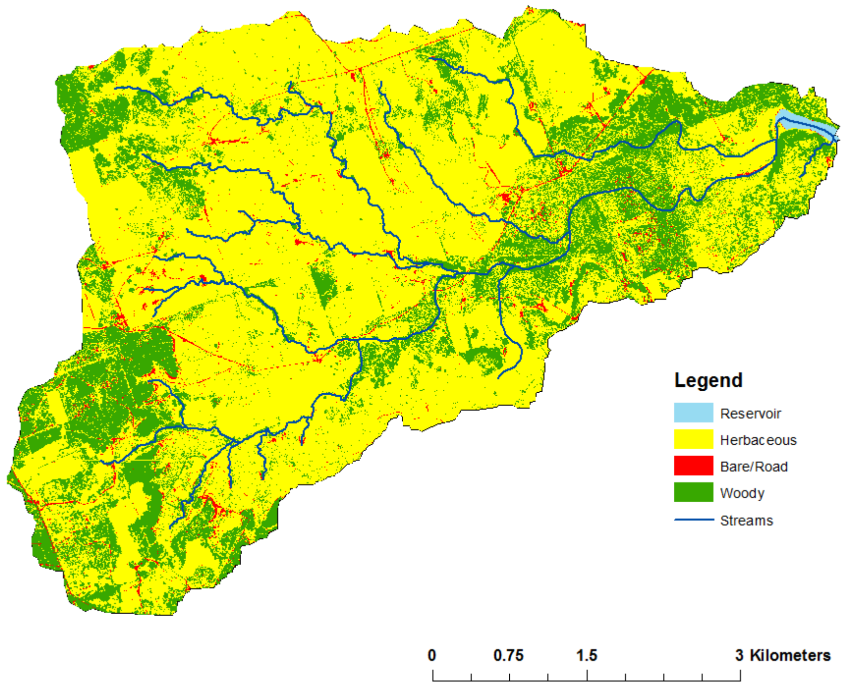
Lampasas Site 3 Classification: 1995



Lampasas Site 3 Cropland: 1995



Lampasas Site 3 Classification: 2004



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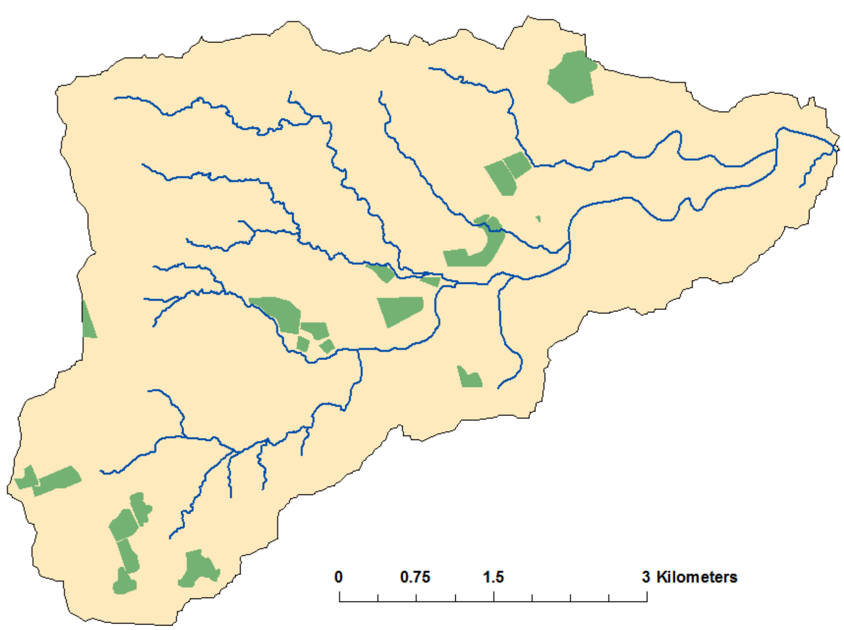

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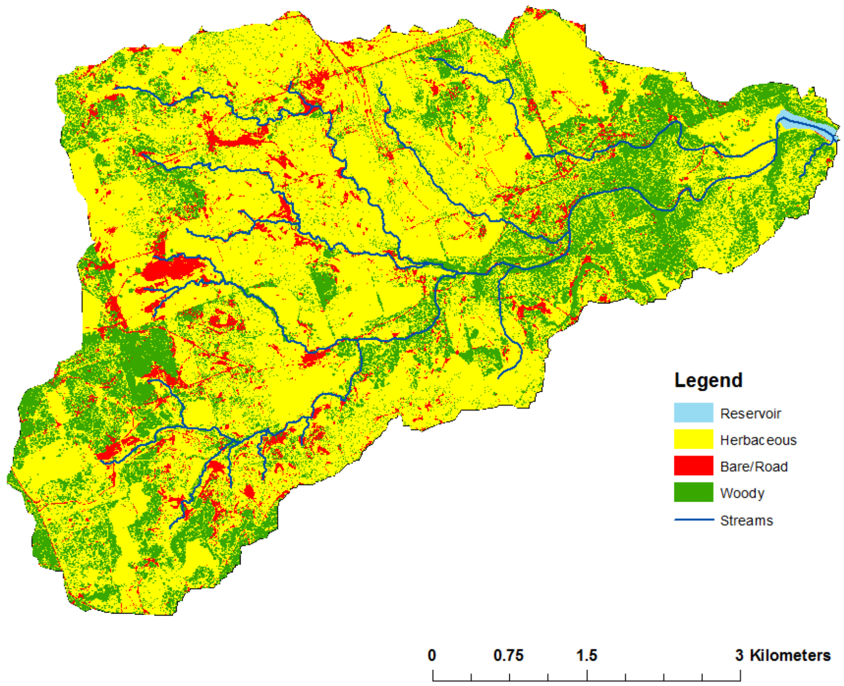

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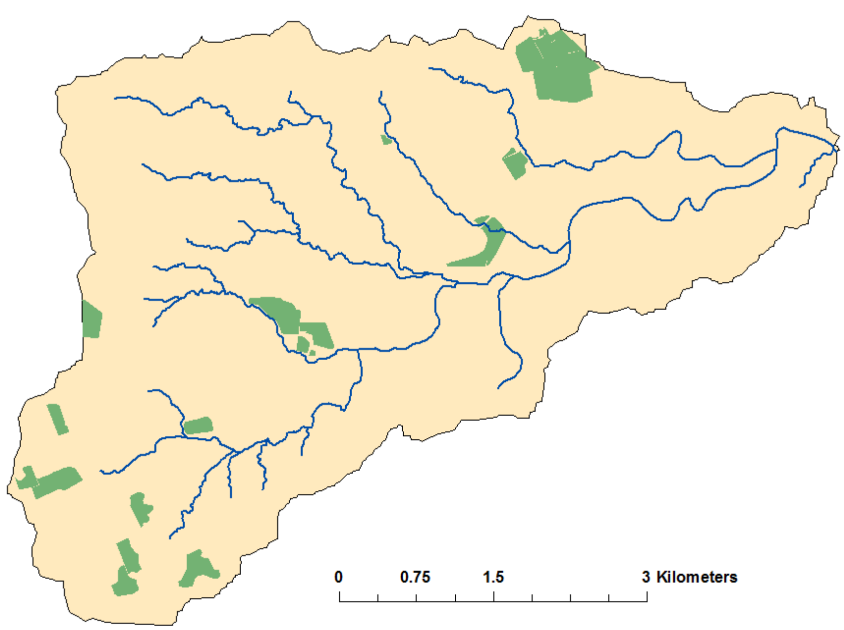
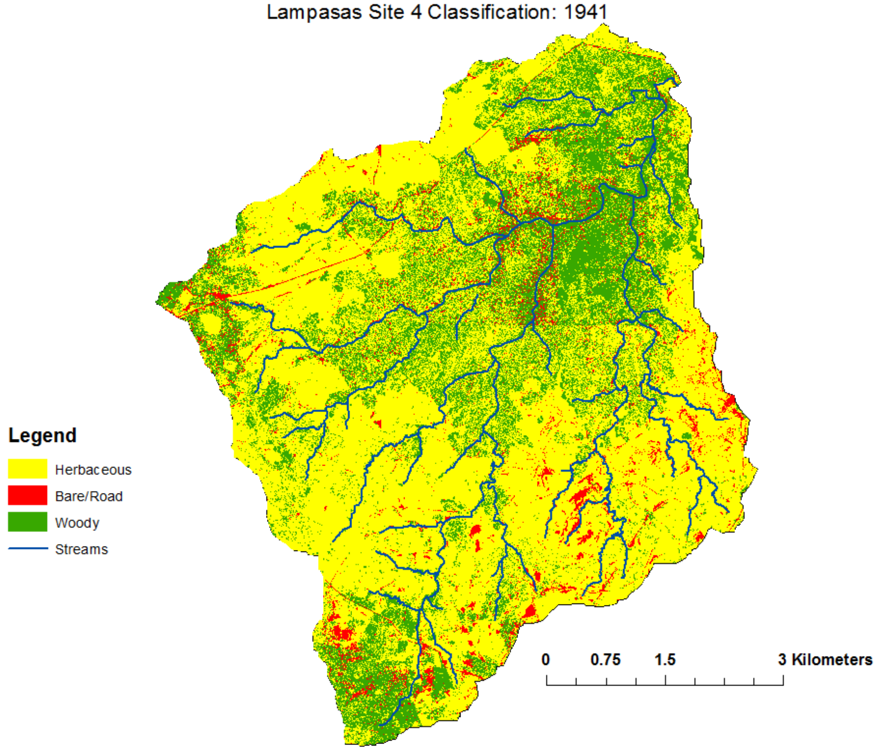
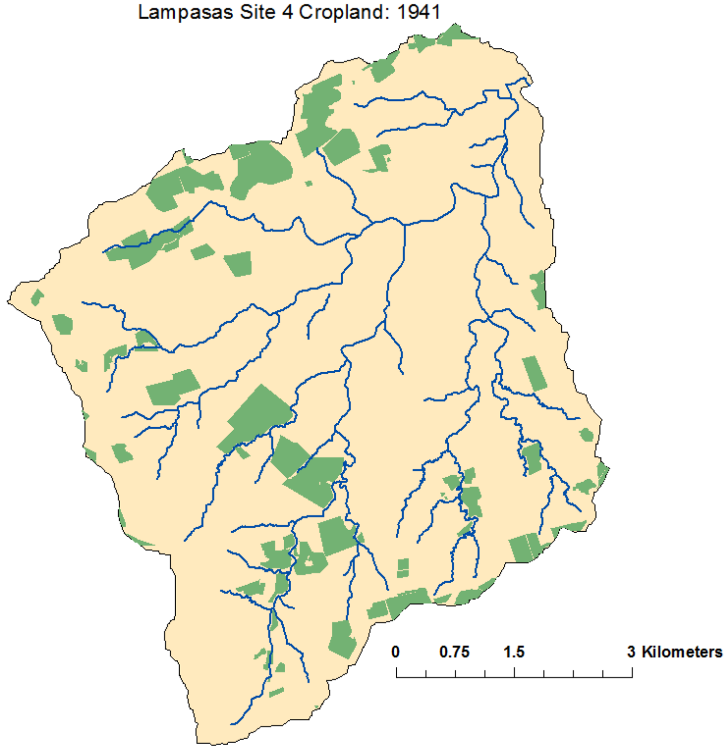


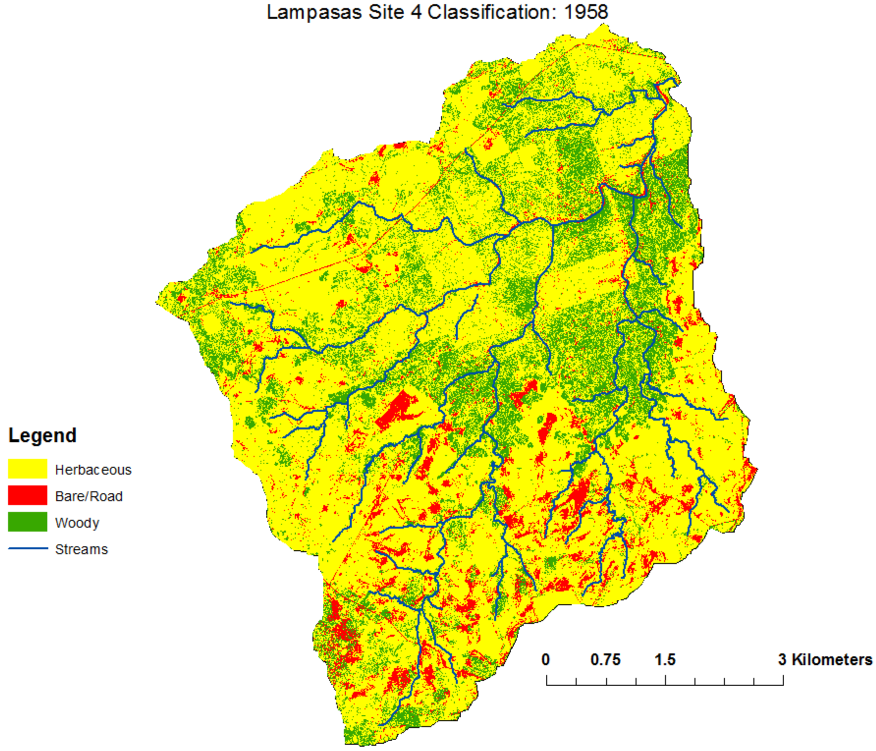
Lampasas Site 4 Classification: 1941



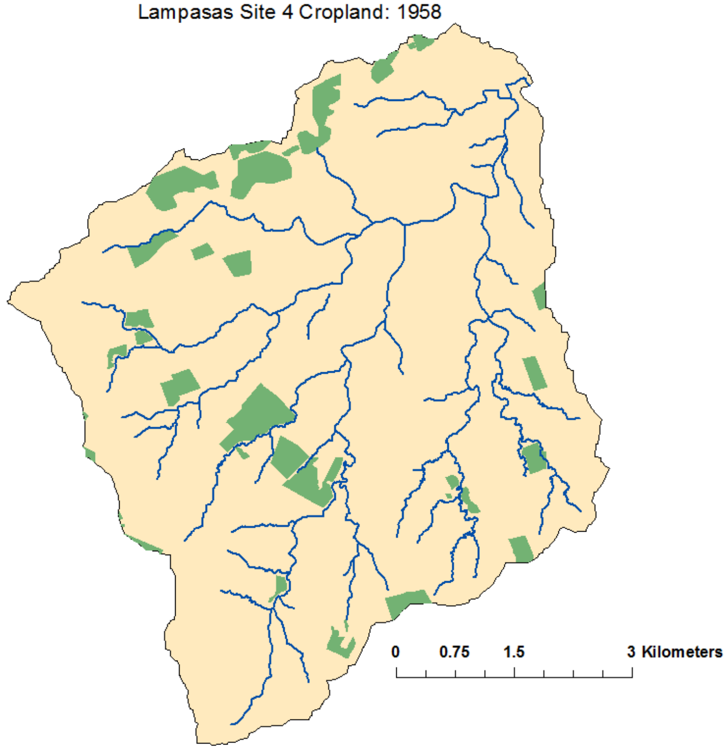
Lampasas Site 4 Cropland: 1941



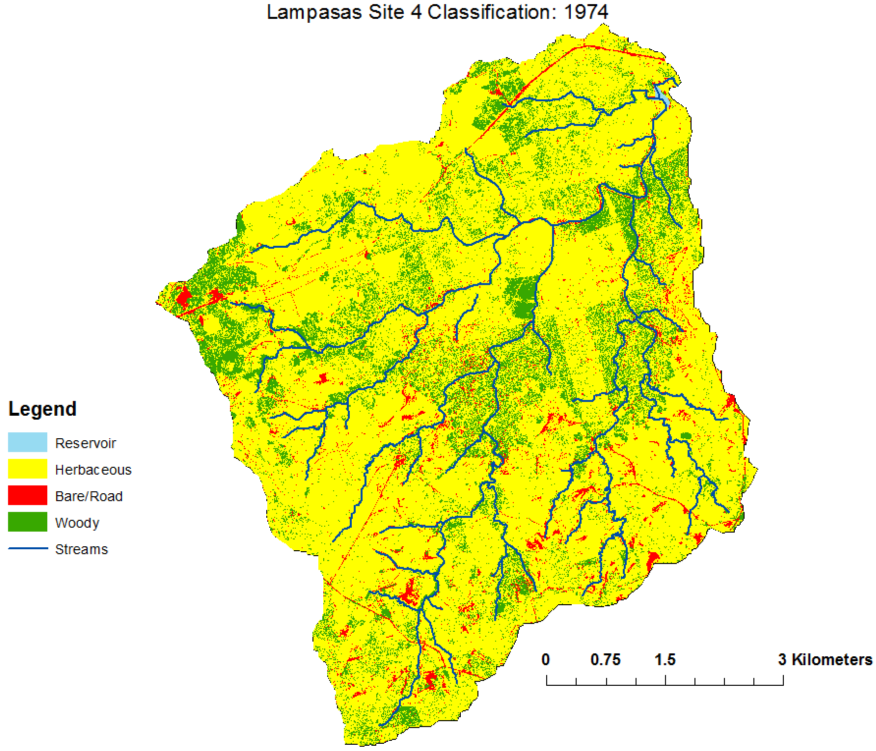
Lampasas Site 4 Classification: 1958



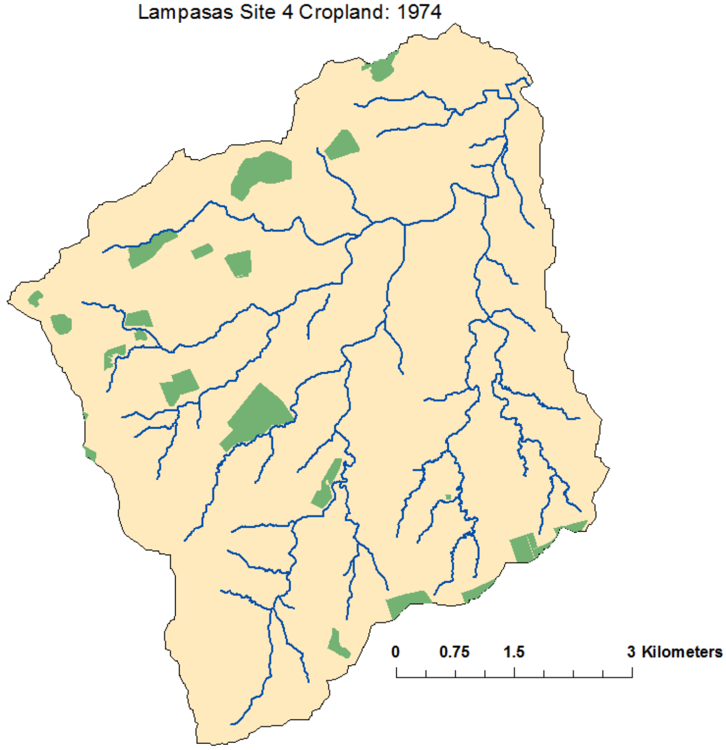
Lampasas Site 4 Cropland: 1958



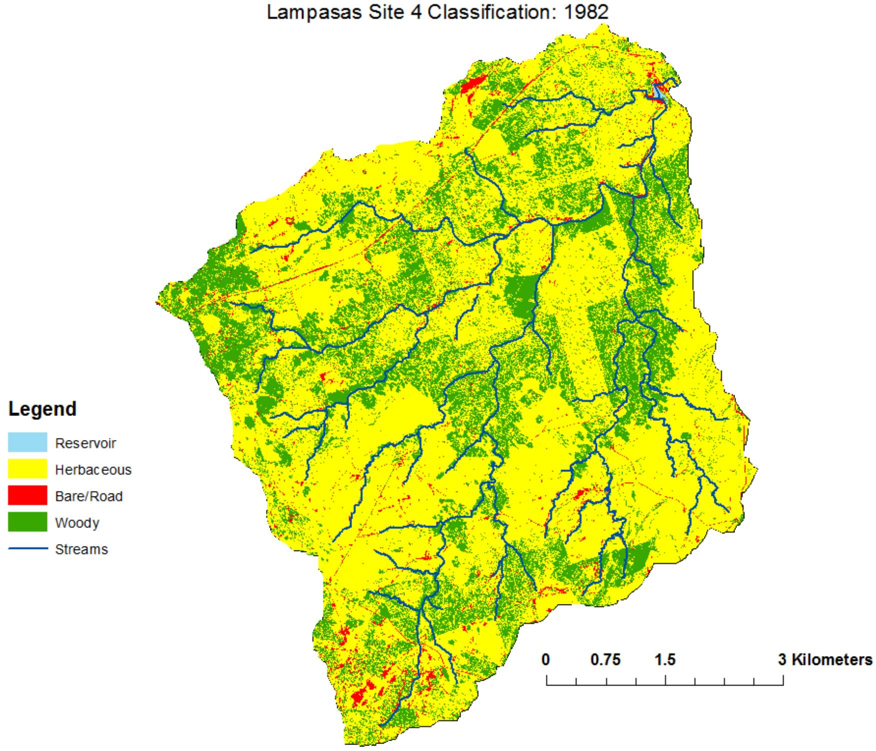
Lampasas Site 4 Classification: 1974



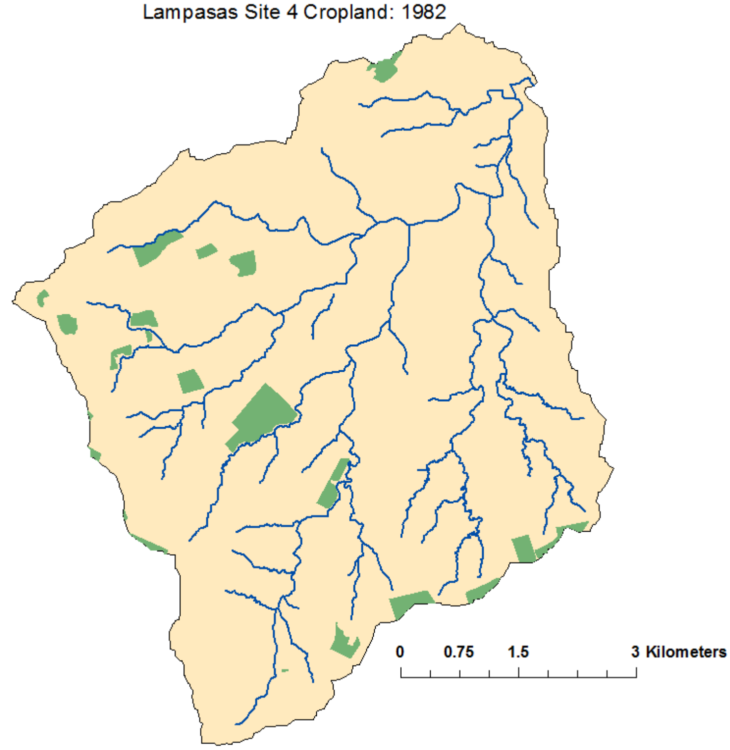
Lampasas Site 4 Cropland: 1974



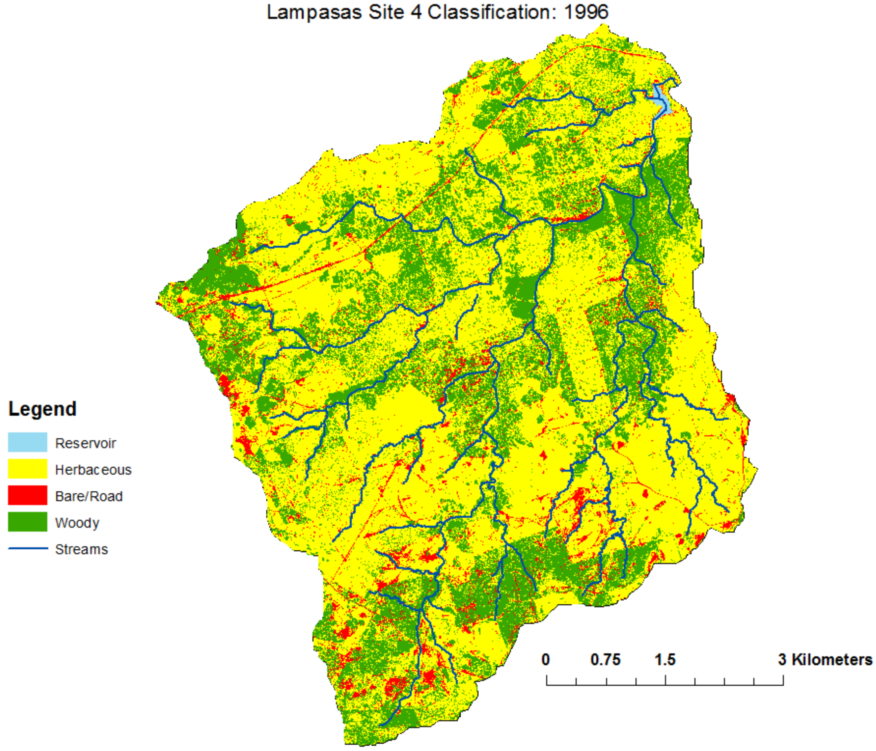
Lampasas Site 4 Classification: 1982



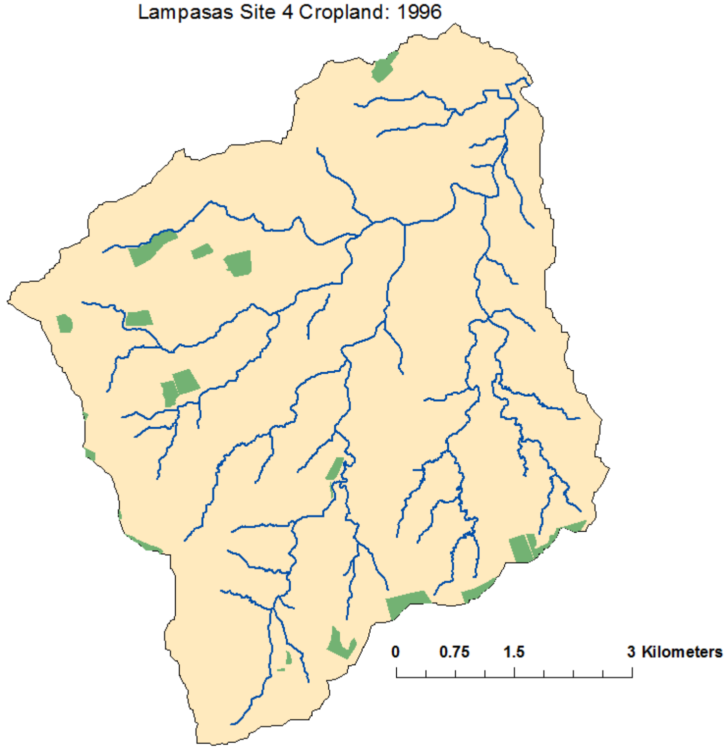
Lampasas Site 4 Cropland: 1982

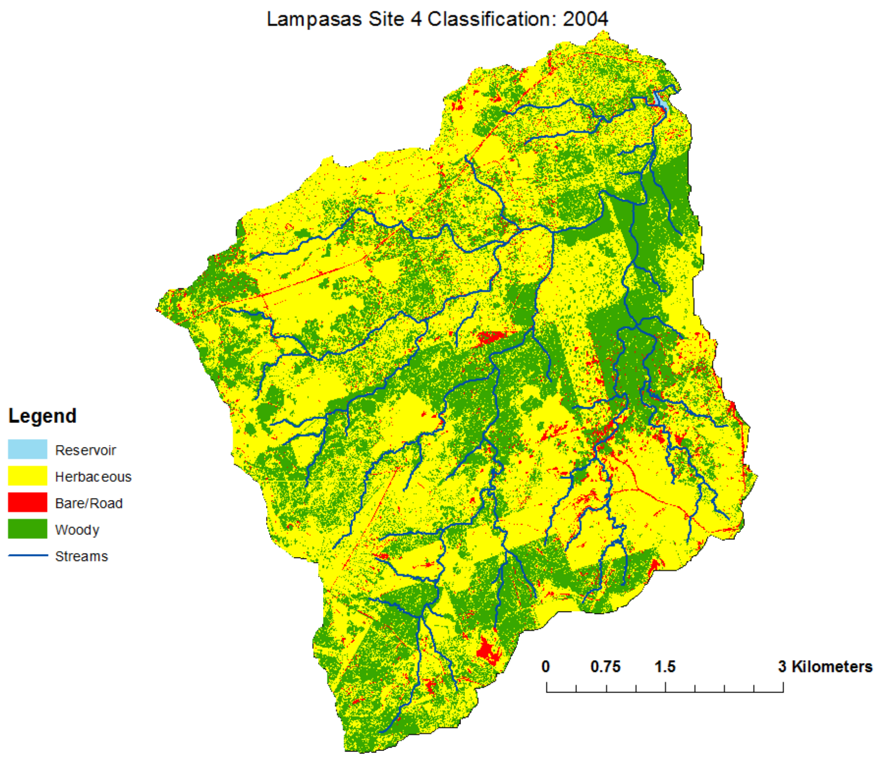


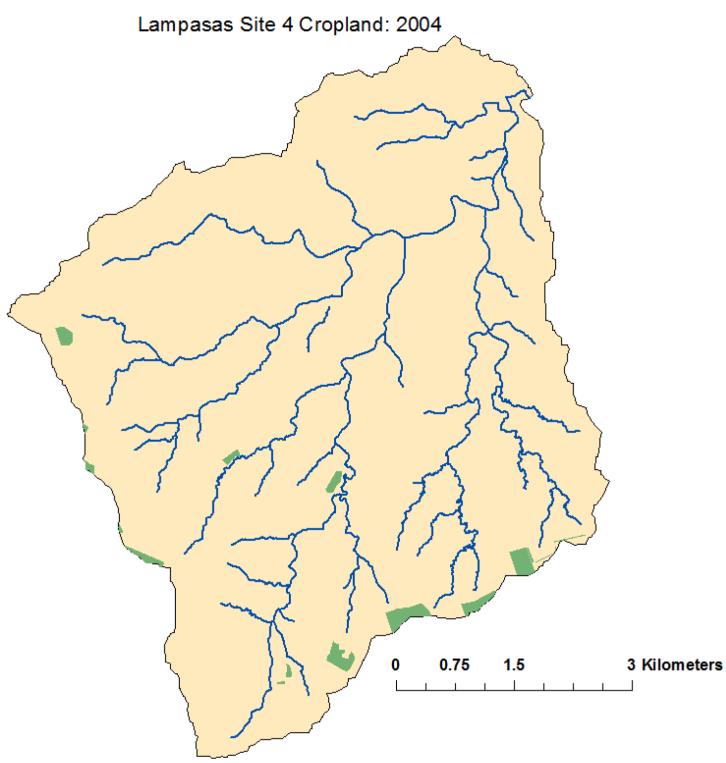
Lampasas Site 4 Classification: 1996



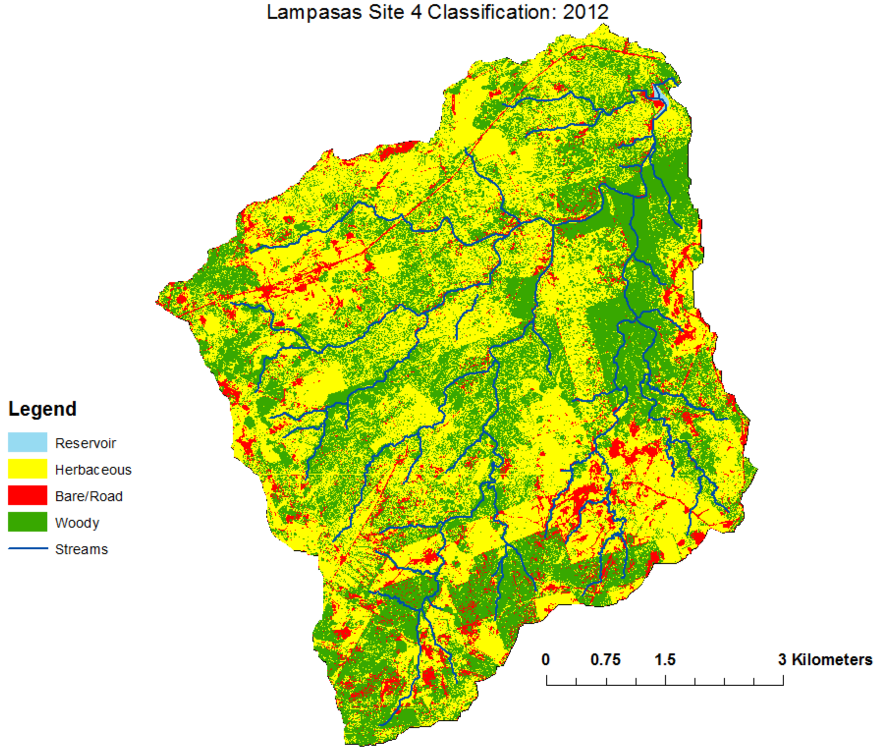
Lampasas Site 4 Cropland: 1996



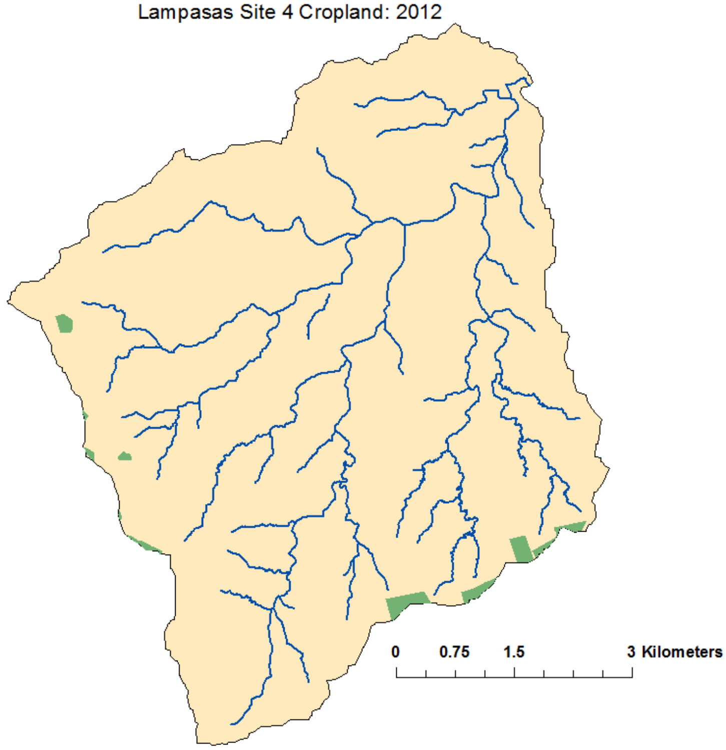

Lampasas Site 4 Classification: 2004
Legend
Reservoir
Herbaceous
Bare/Road
Woody
Streams
0
0.75
1.5
3 Kilometers


Lampasas Site 4 Cropland: 2004
0
0.75
1.5
3 Kilometers

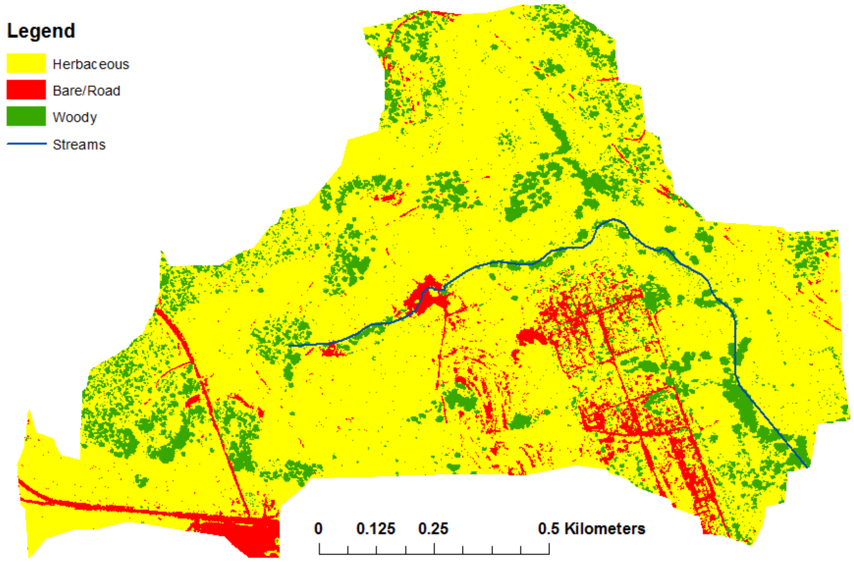
Lampasas Site 4 Classification: 2012



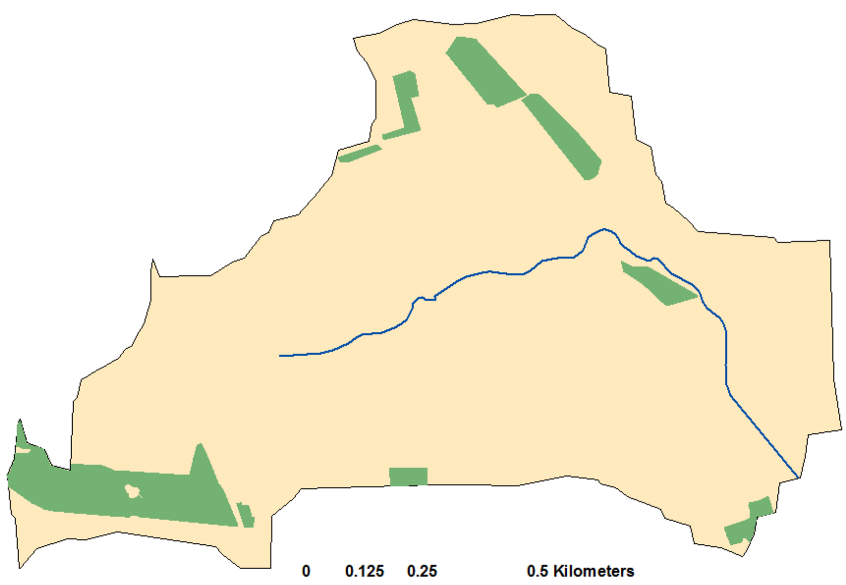
Lampasas Site 4 Cropland: 2012



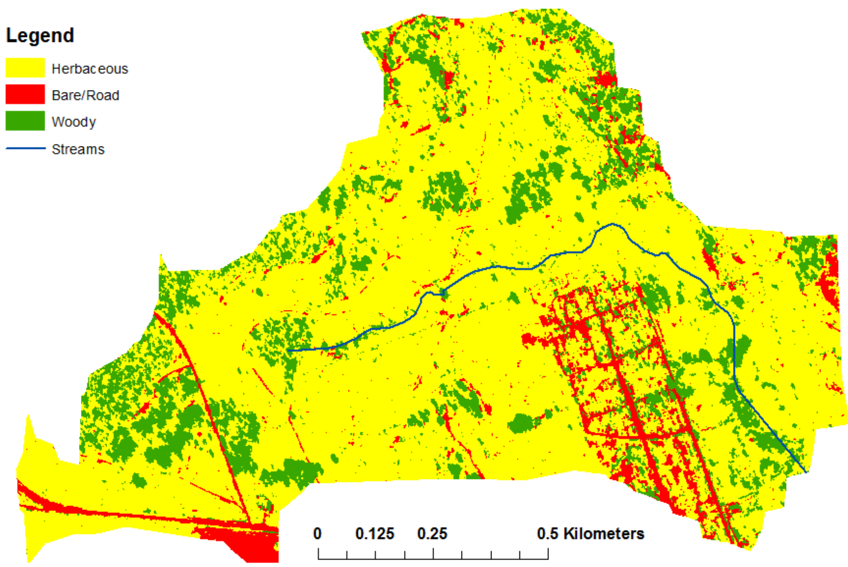
Lampasas Site 9 Classification: 1941



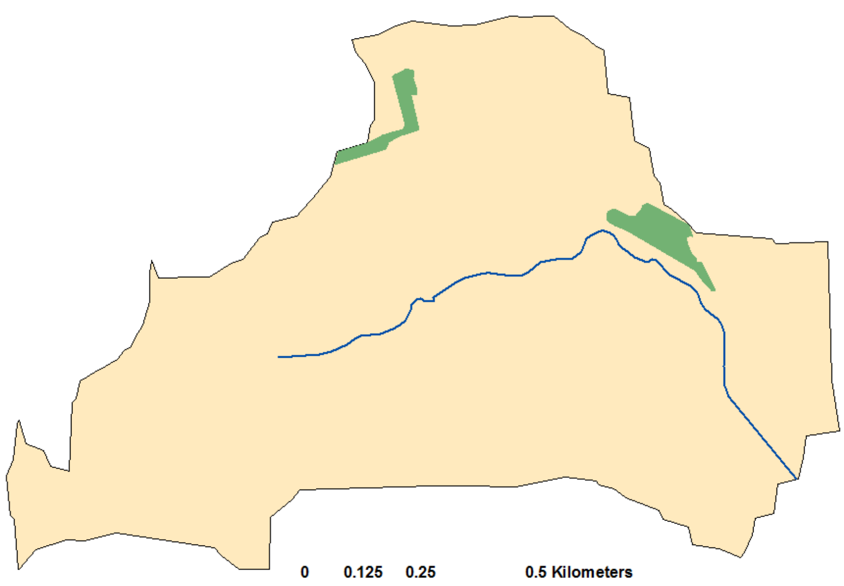
Lampasas Site 9 Cropland: 1941



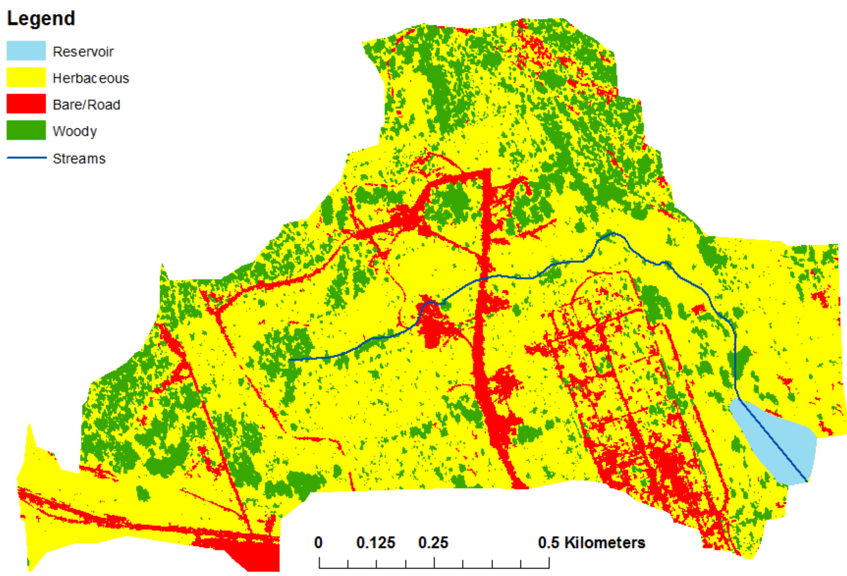
Lampasas Site 9 Classification: 1958



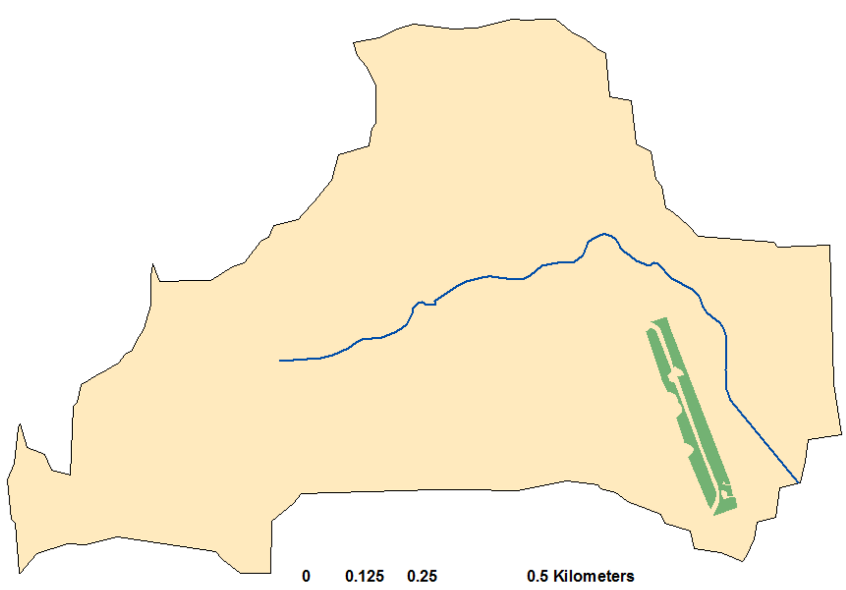
Lampasas Site 9 Cropland: 1958



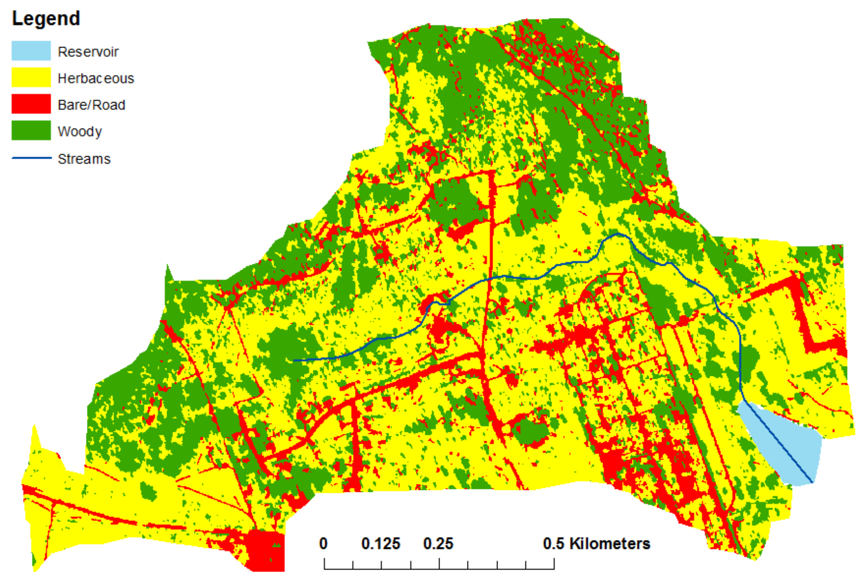
Lampasas Site 9 Classification: 1974



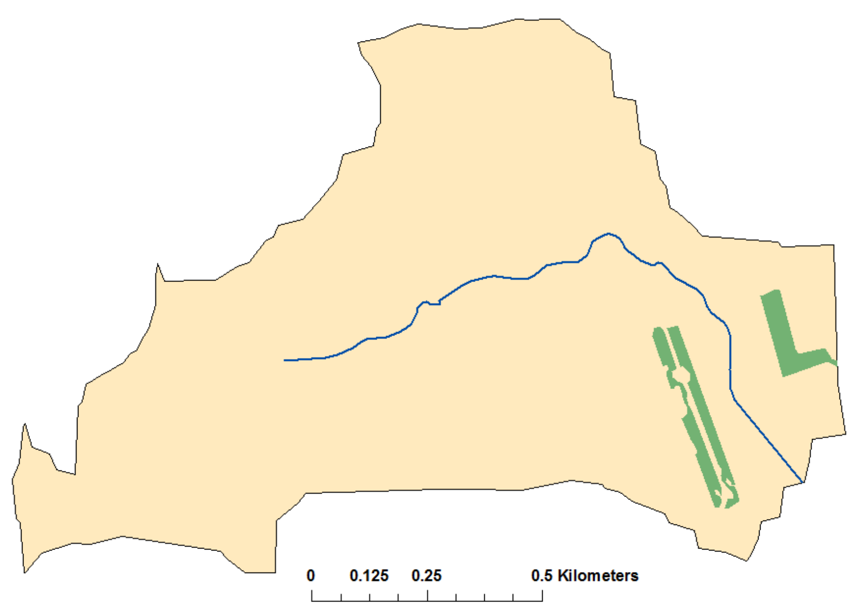
Lampasas Site 9 Cropland: 1974



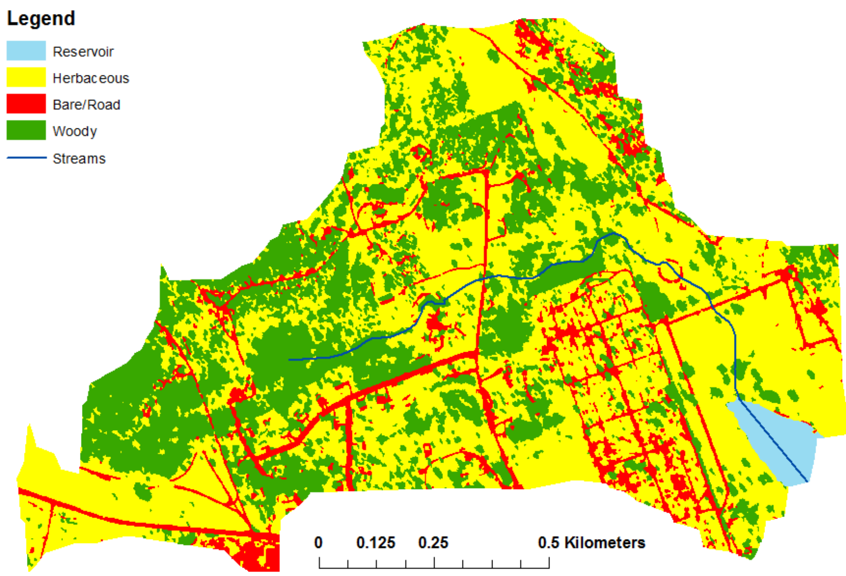
Lampasas Site 9 Classification: 1982



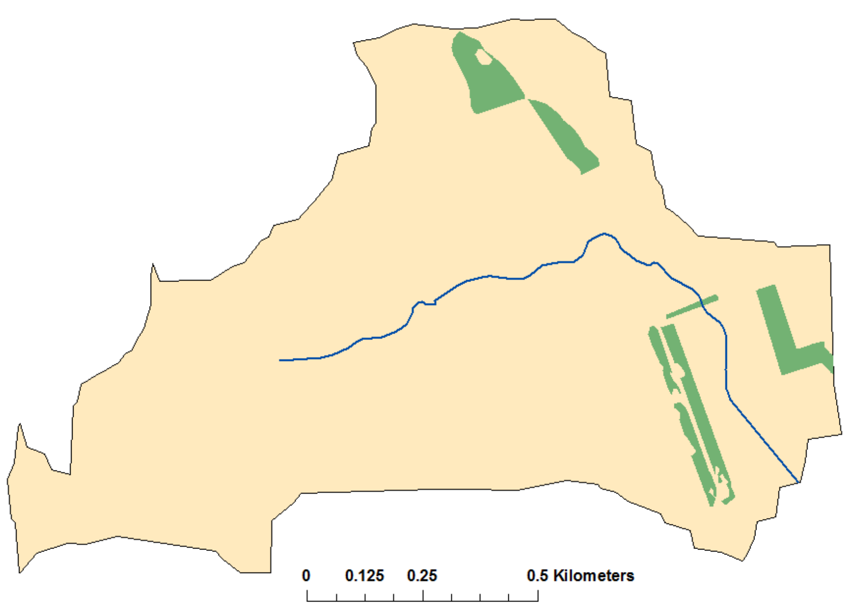
Lampasas Site 9 Cropland: 1982



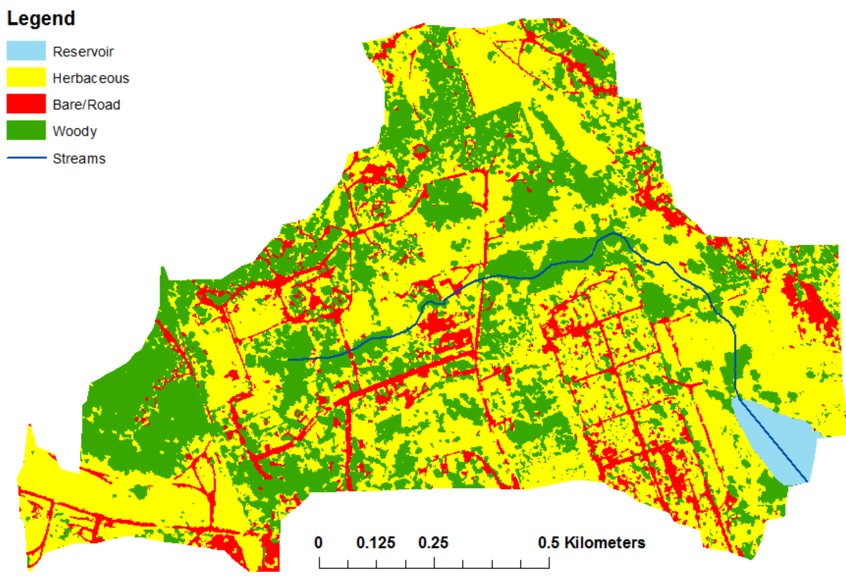
Lampasas Site 9 Classification: 1996



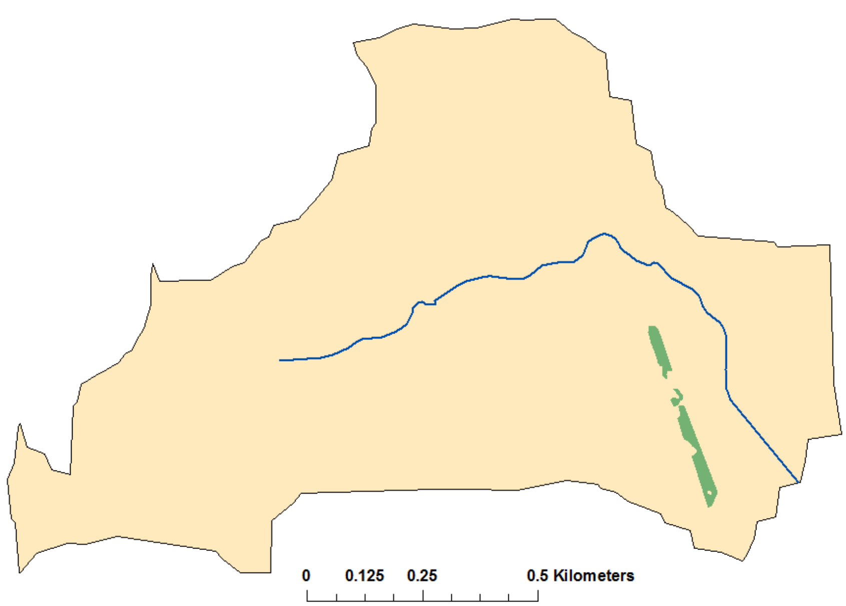
Lampasas Site 9 Cropland: 1996



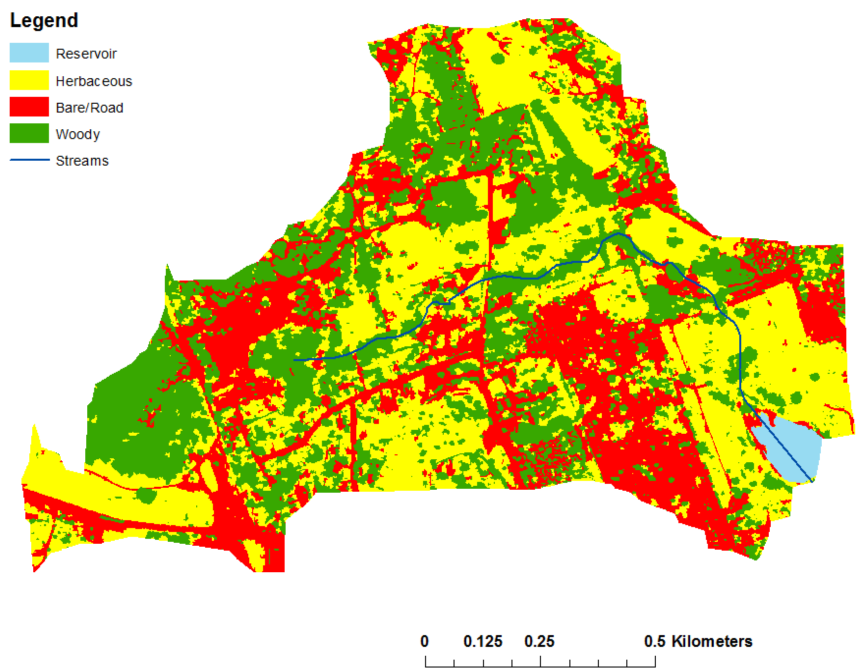
Lampasas Site 9 Classification: 2004



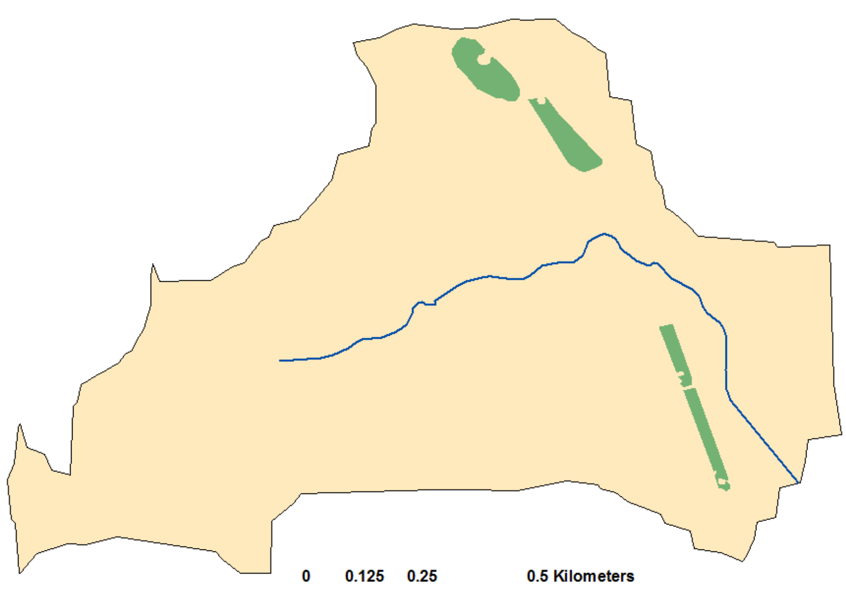
Lampasas Site 9 Cropland: 2004



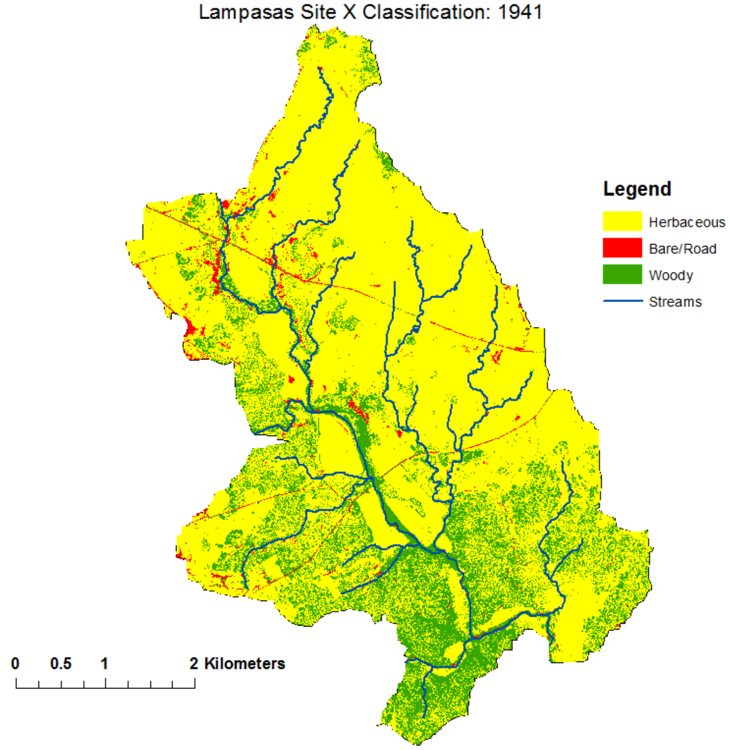
Lampasas Site 9 Classification: 2012



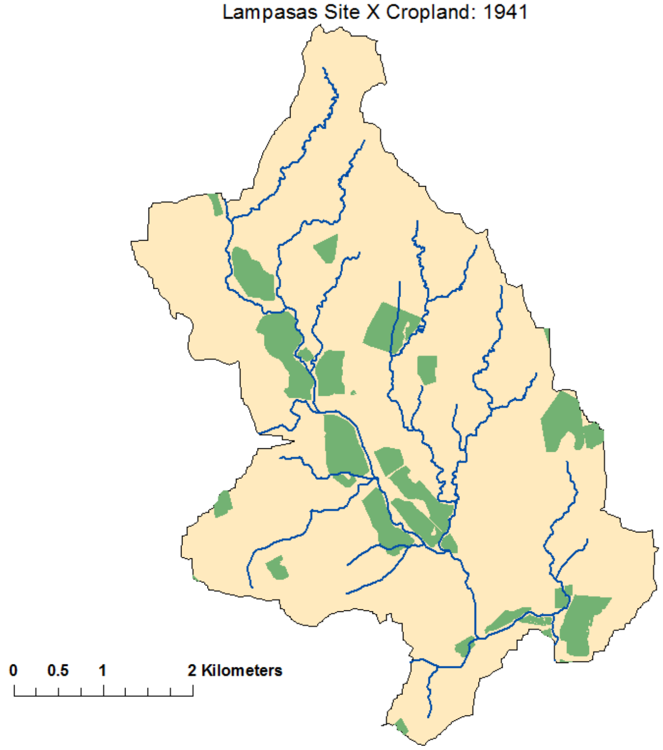
Lampasas Site 9 Cropland: 2012



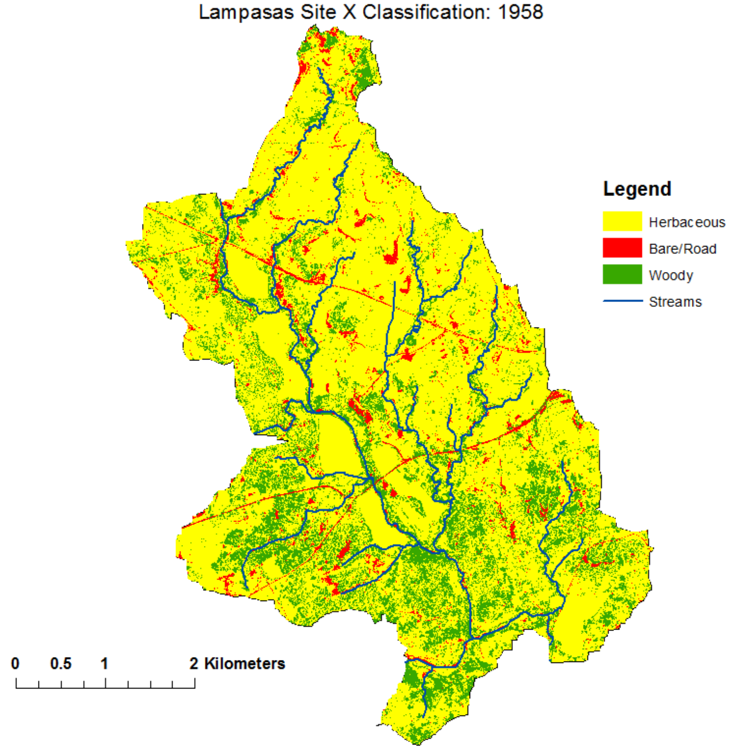
Lampasas Site X Classification: 1941



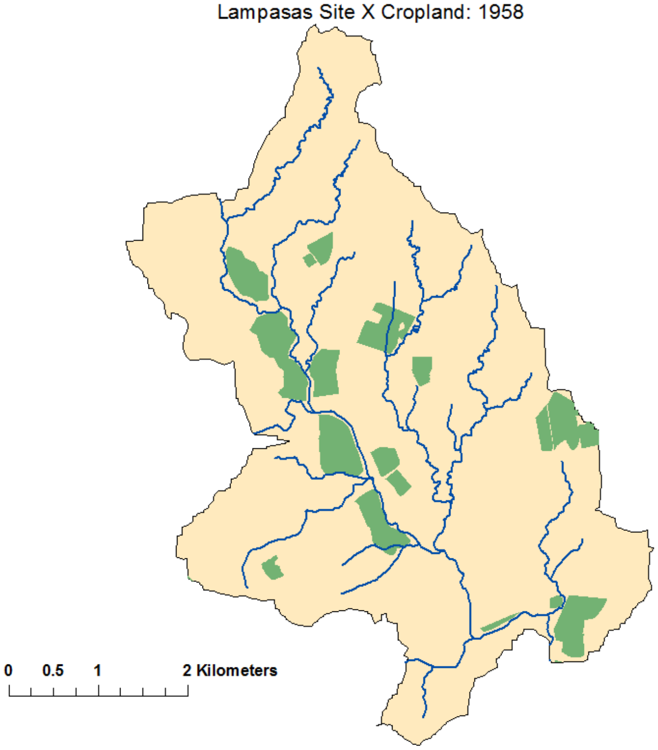
Lampasas Site X Cropland: 1941

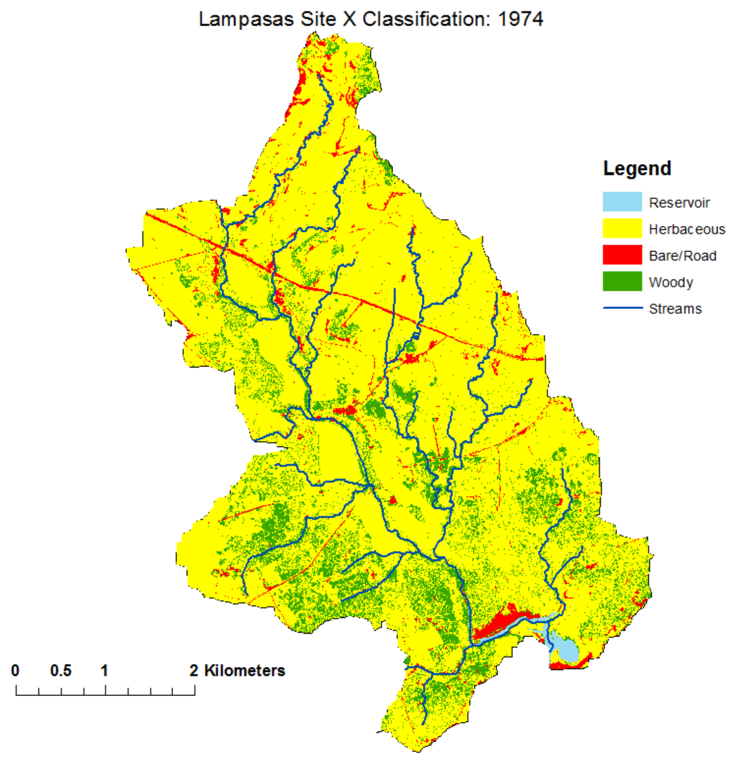


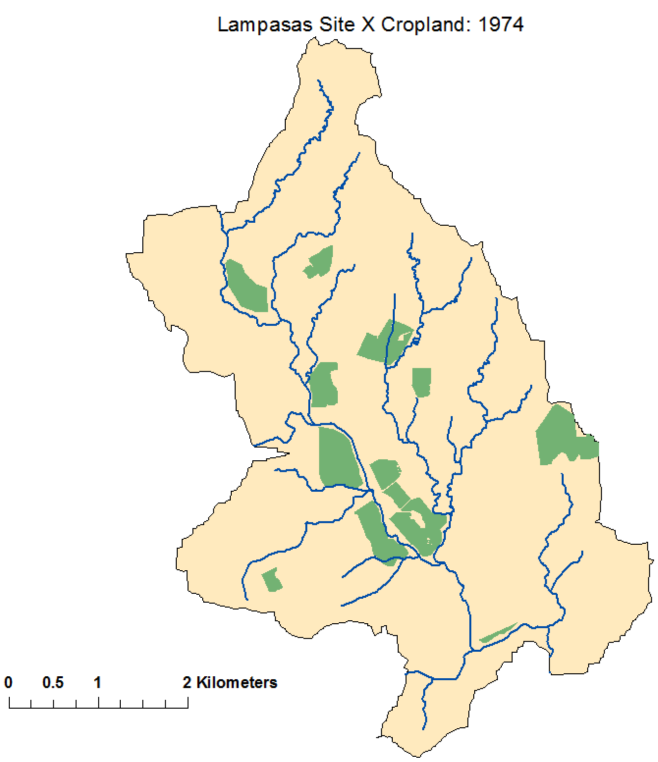
Lampasas Site X Classification: 1958



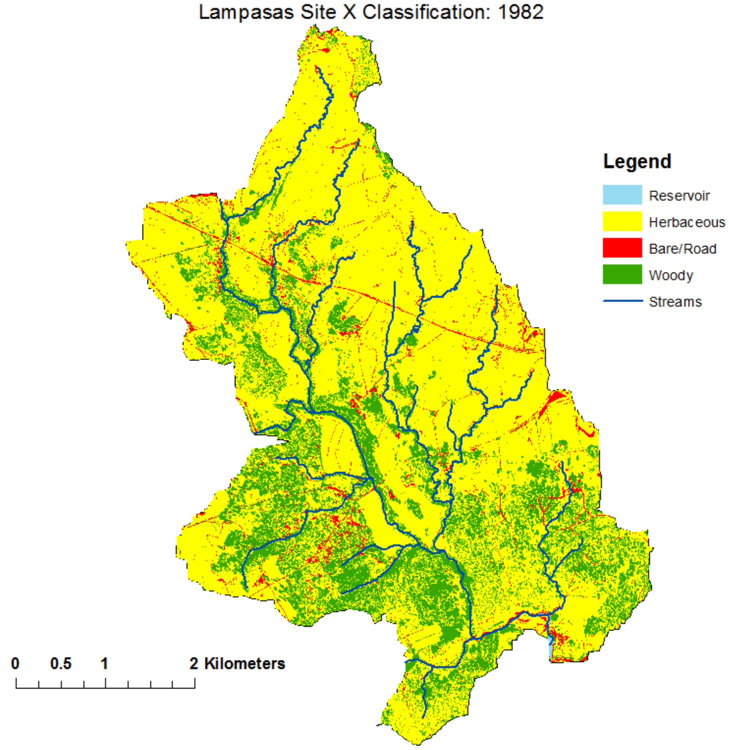
Lampasas Site X Cropland: 1958



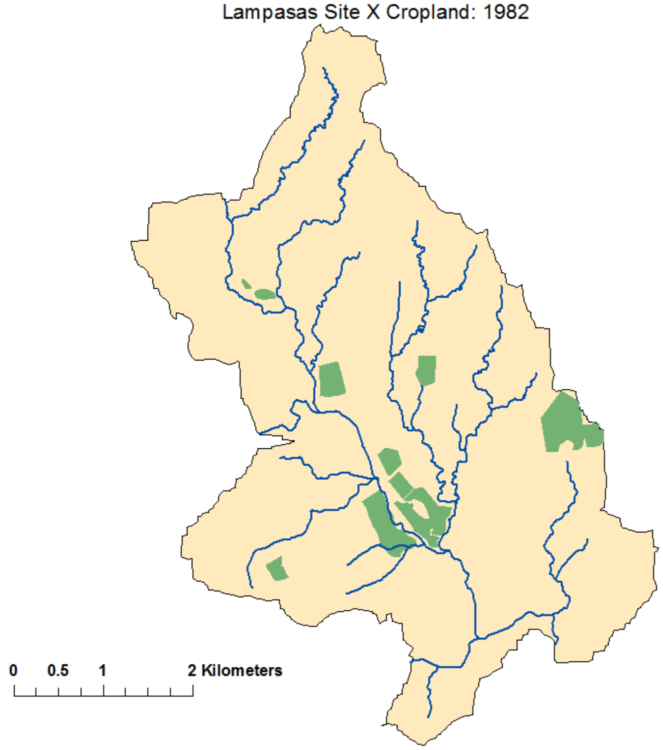

Lampasas Site X Classification: 1974
Legend
Reservoir
Herbaceous
Bare/Road
Woody
Streams
0 0.5 1 2 Kilometers


Lampasas Site X Cropland: 1974
0 0.5 1 2 Kilometers

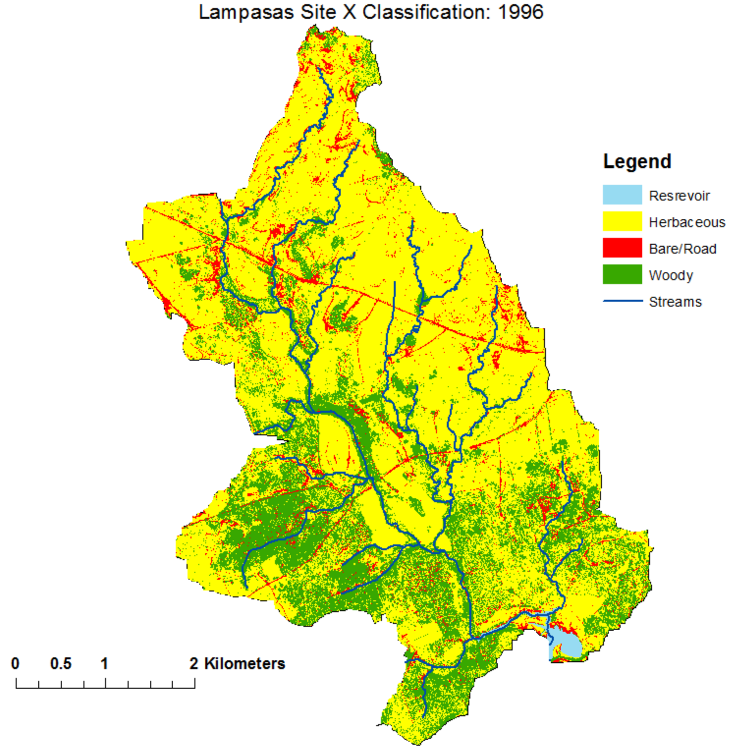
Lampasas Site X Classification: 1982



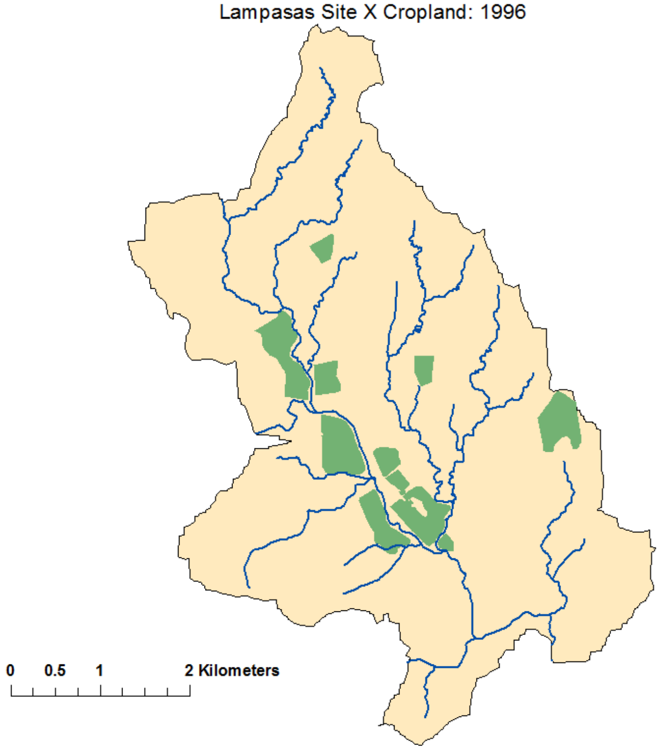
Lampasas Site X Cropland: 1982



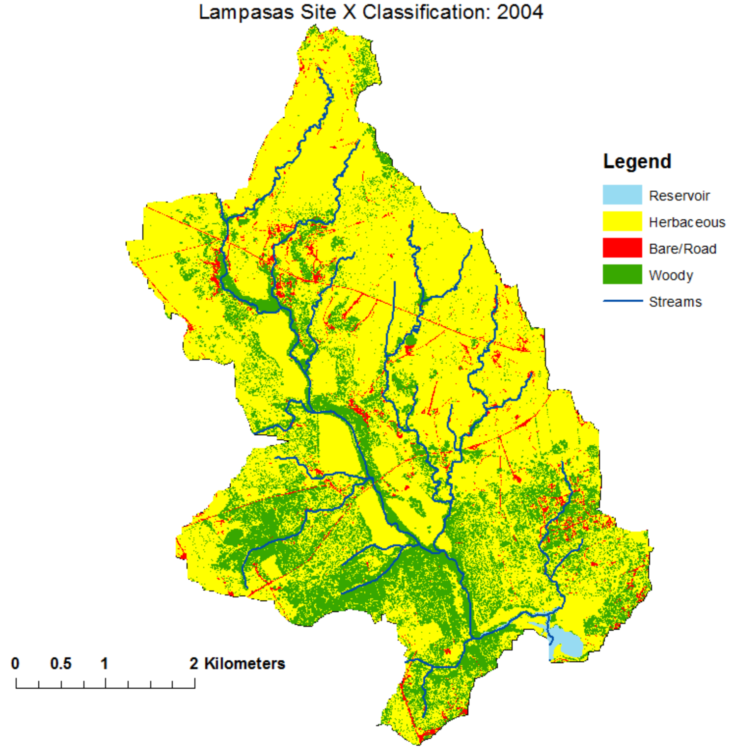
Lampasas Site X Classification: 1996



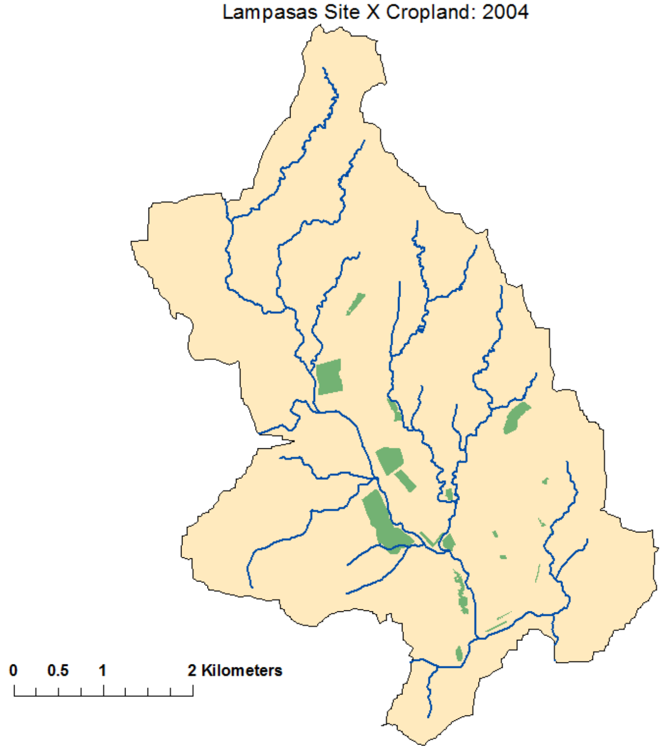
Lampasas Site X Cropland: 1996



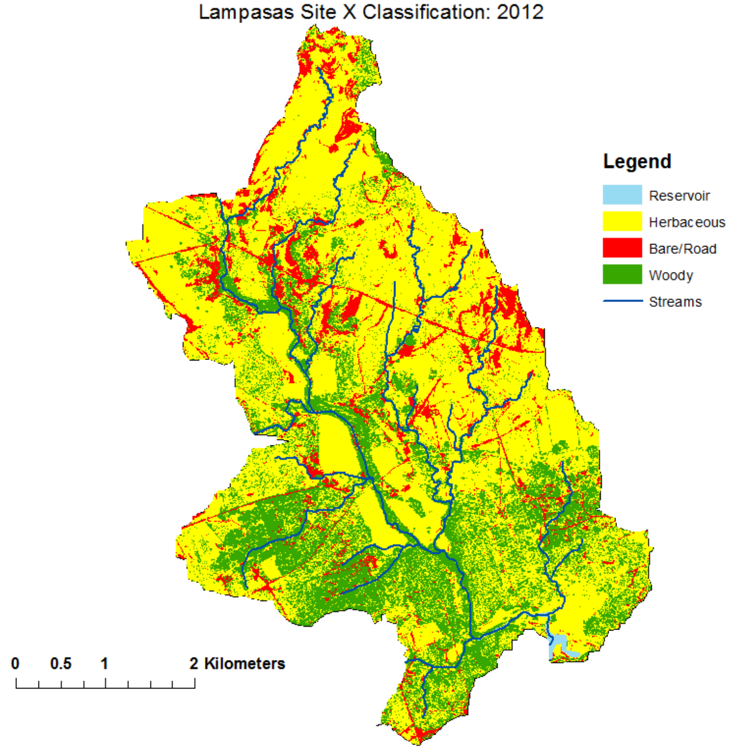
Lampasas Site X Classification: 2004



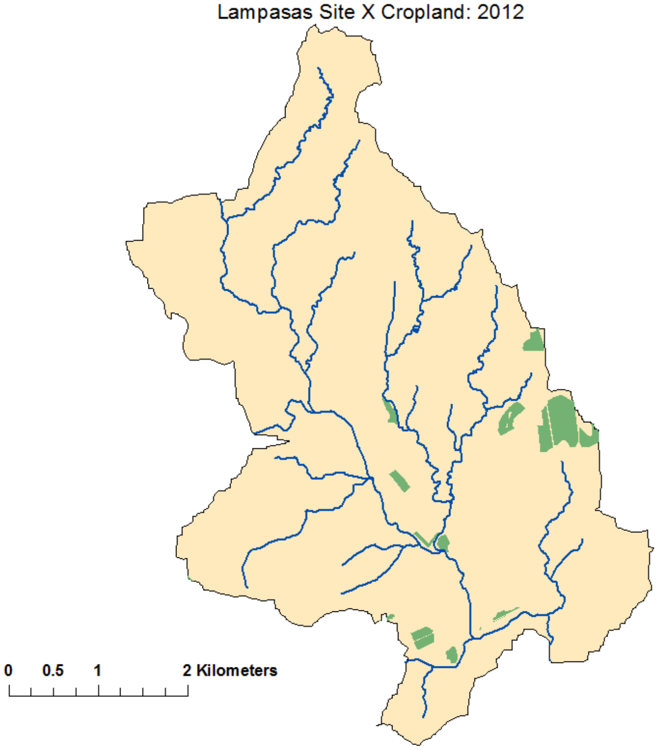
Lampasas Site X Cropland: 2004



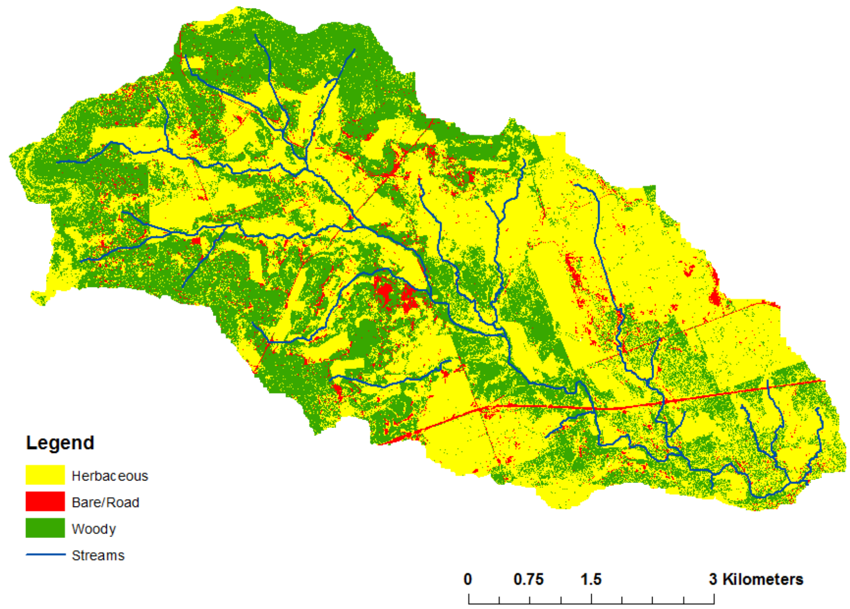
Lampasas Site X Classification: 2012



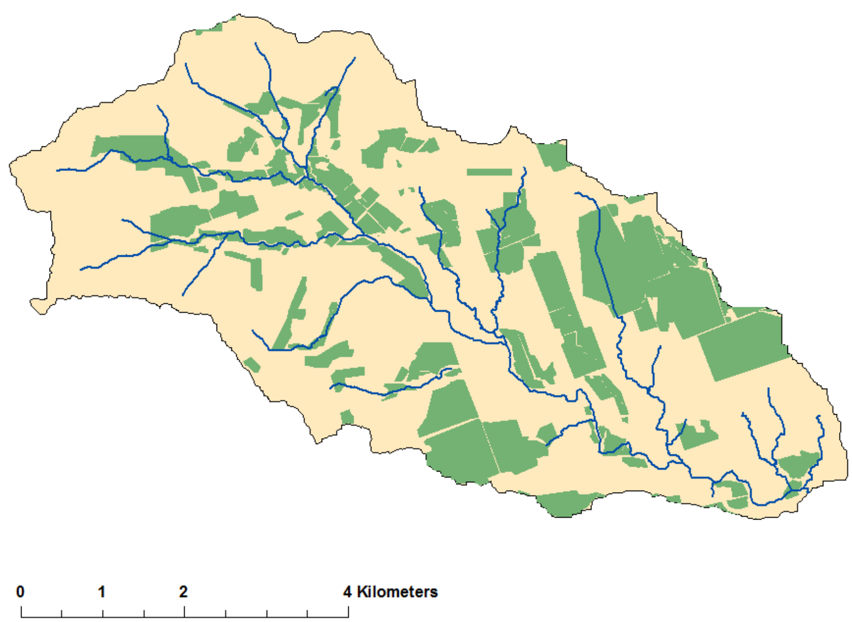
Lampasas Site X Cropland: 2012



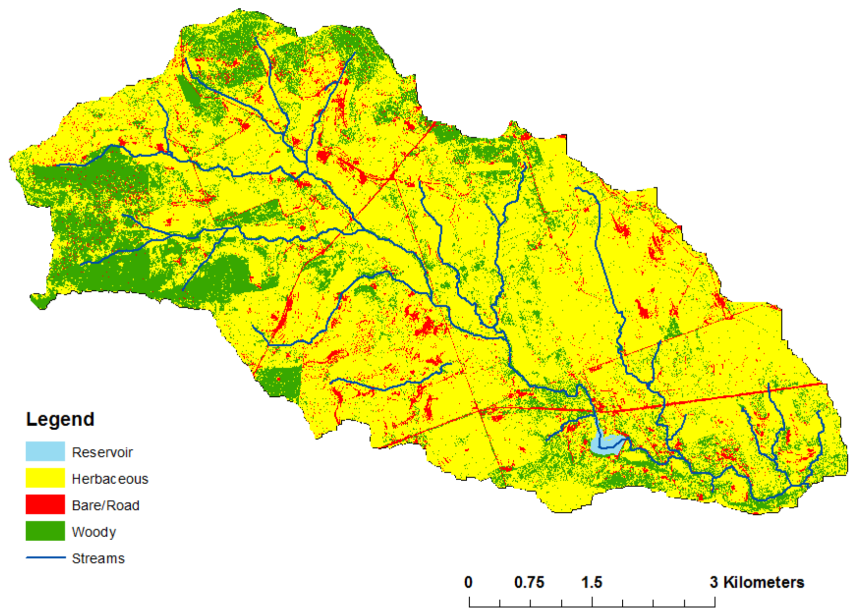
Mills Site 1 Classification: 1937



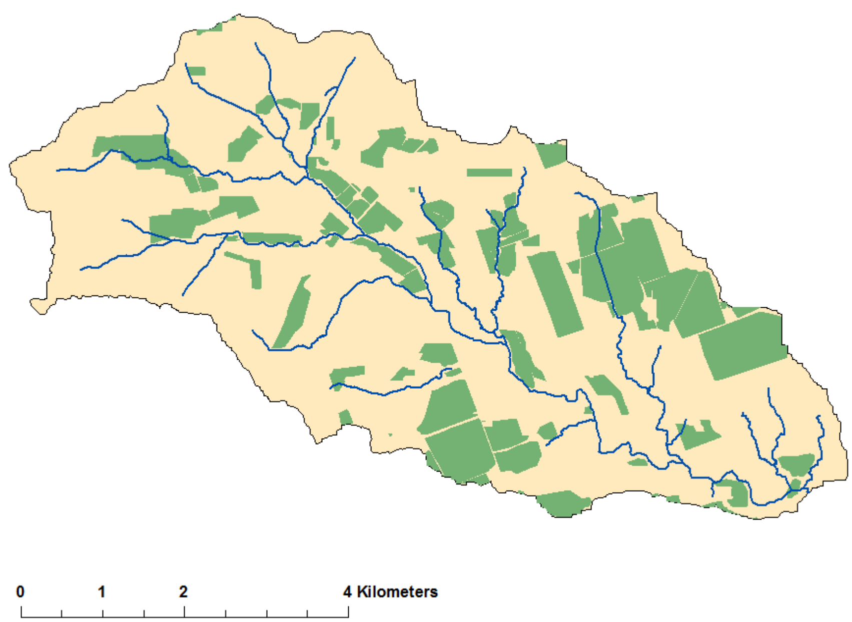
Mills Site 1 Cropland: 1937



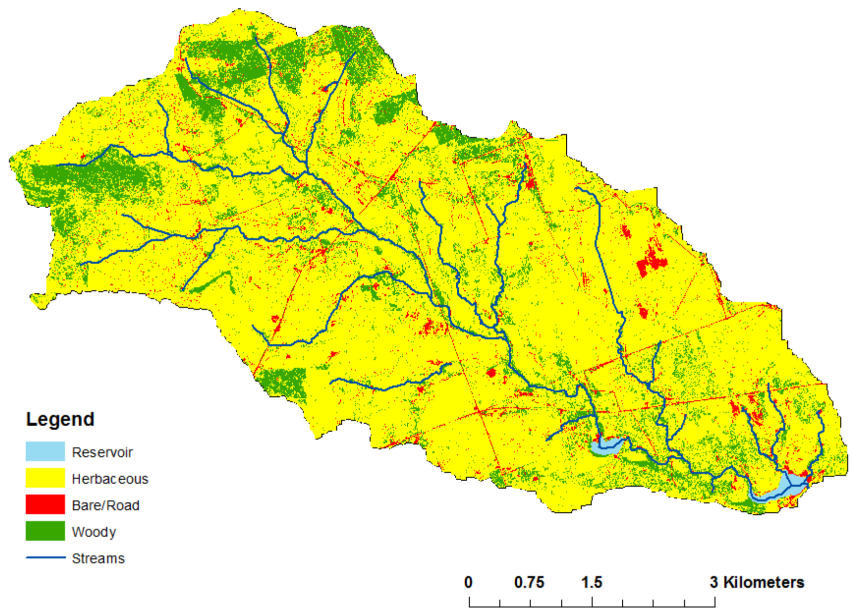
Mills Site 1 Classification: 1958



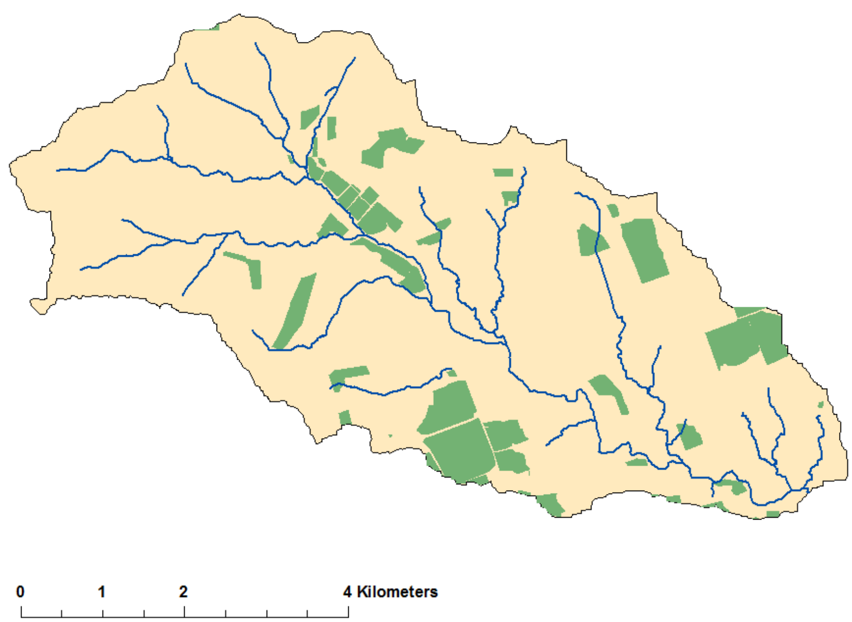
Mills Site 1 Cropland: 1958



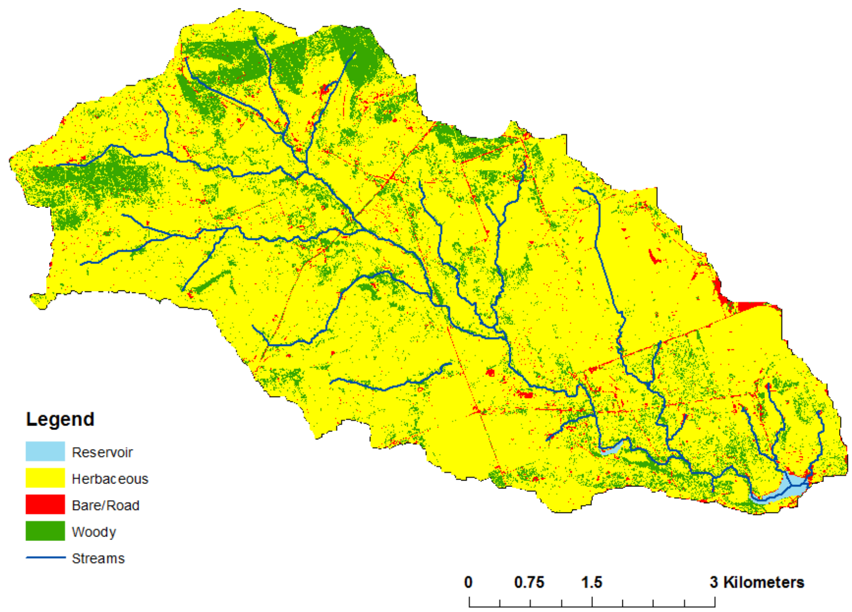
Mills Site 1 Classification: 1975



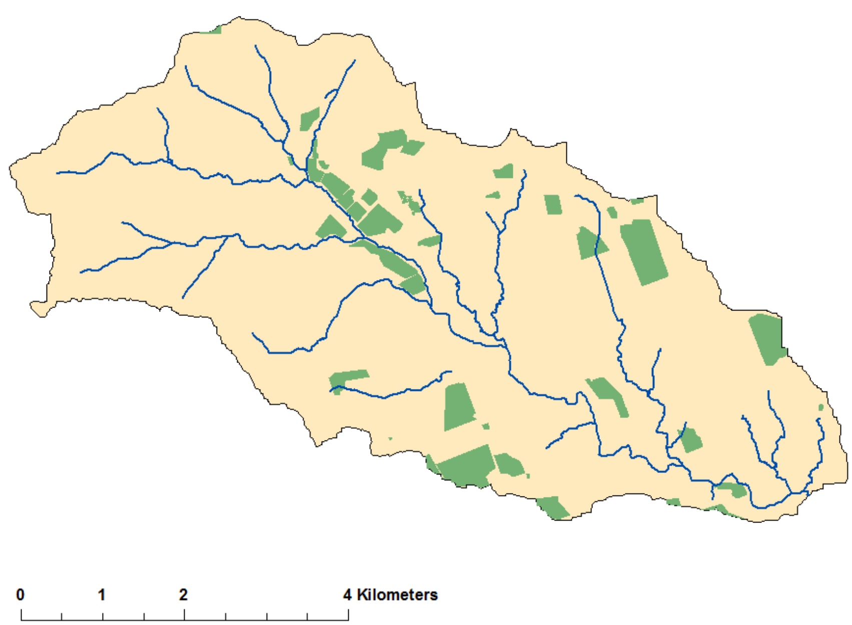
Mills Site 1 Cropland: 1975



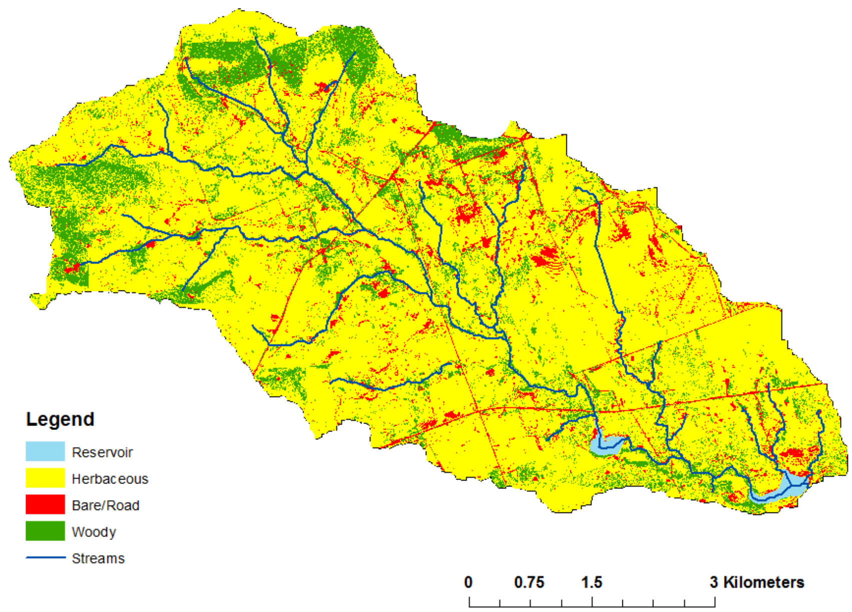
Mills Site 1 Classification: 1982



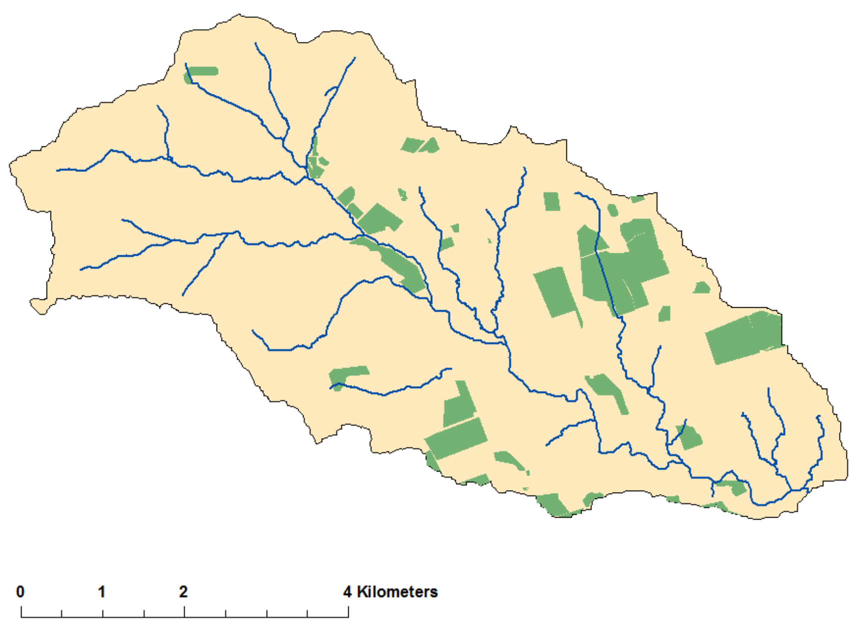
Mills Site 1 Cropland: 1982



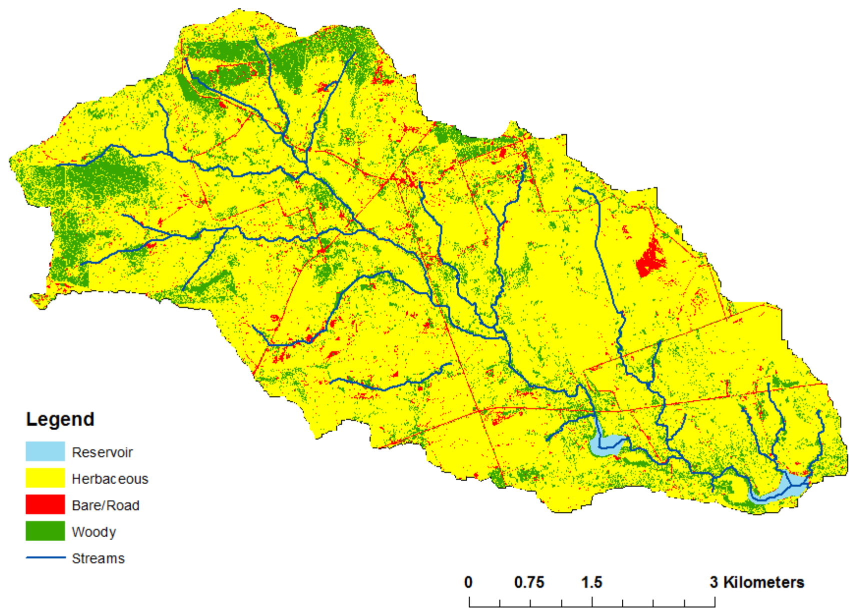
Mills Site 1 Classification: 1995



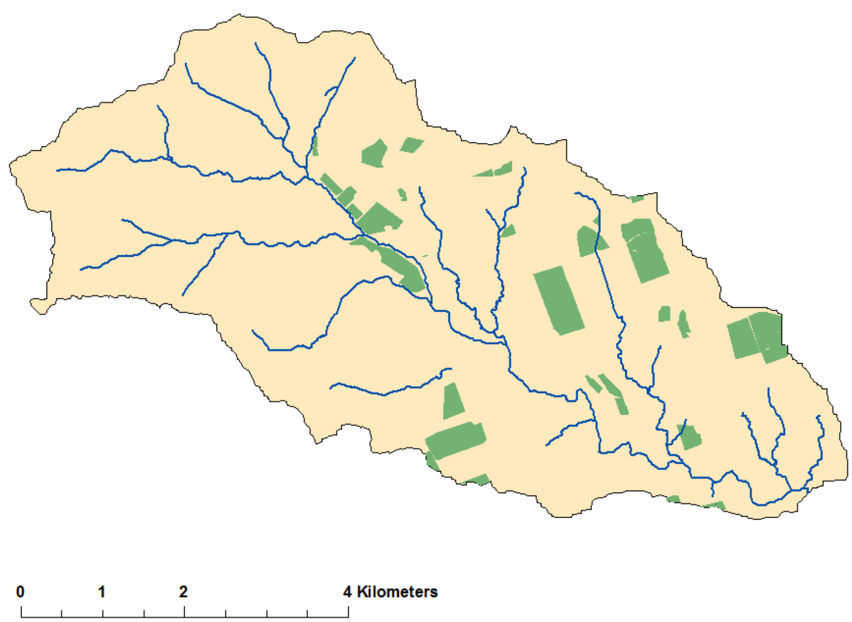
Mills Site 1 Cropland: 1995



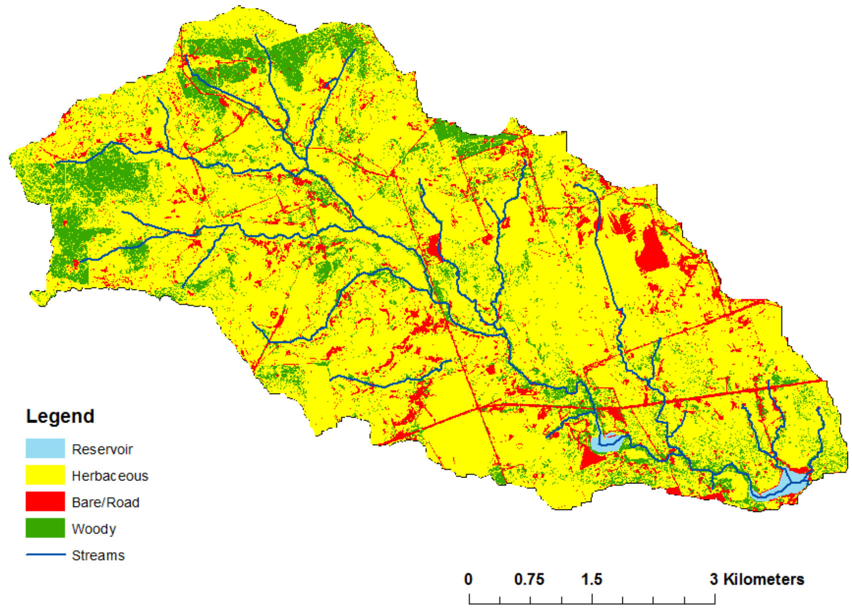
Mills Site 1 Classification: 2004



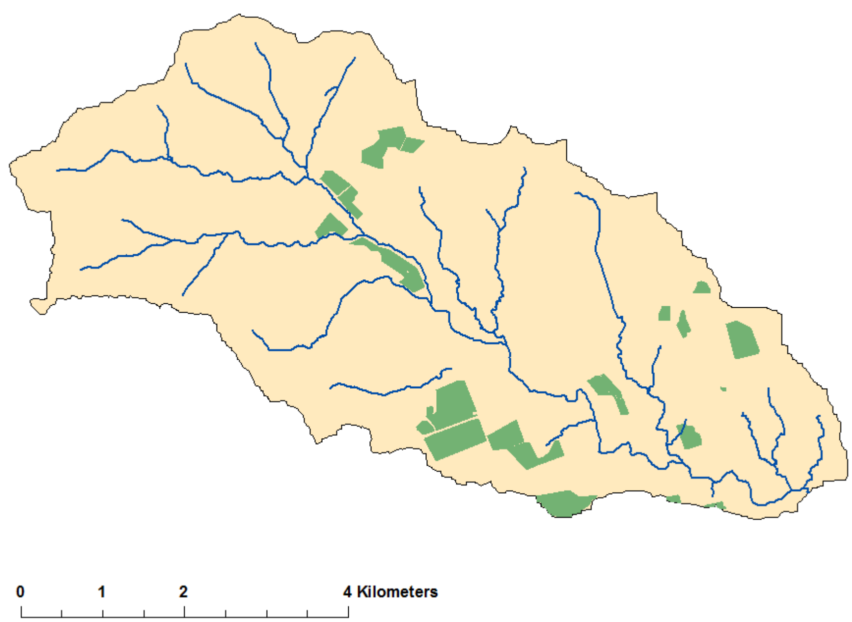
Mills Site 1 Cropland: 2004



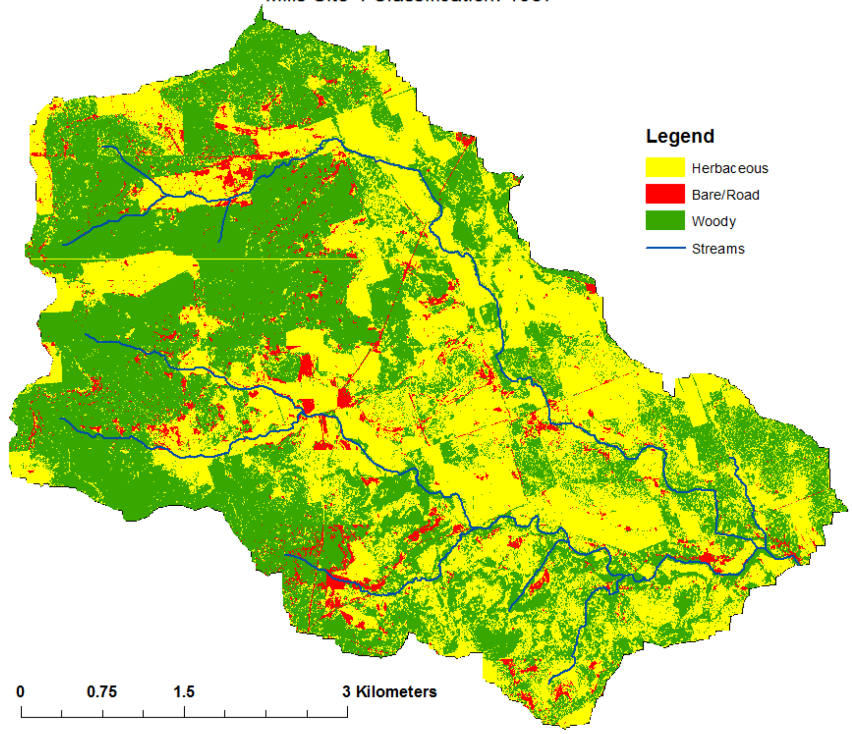
Mills Site 1 Classification: 2012



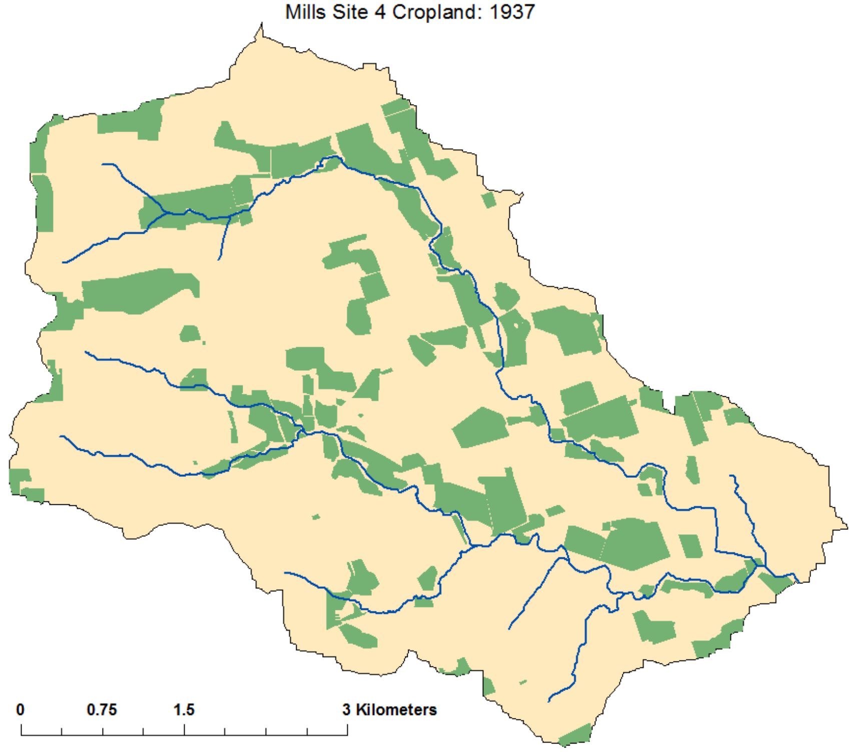
Mills Site 1 Cropland: 2012

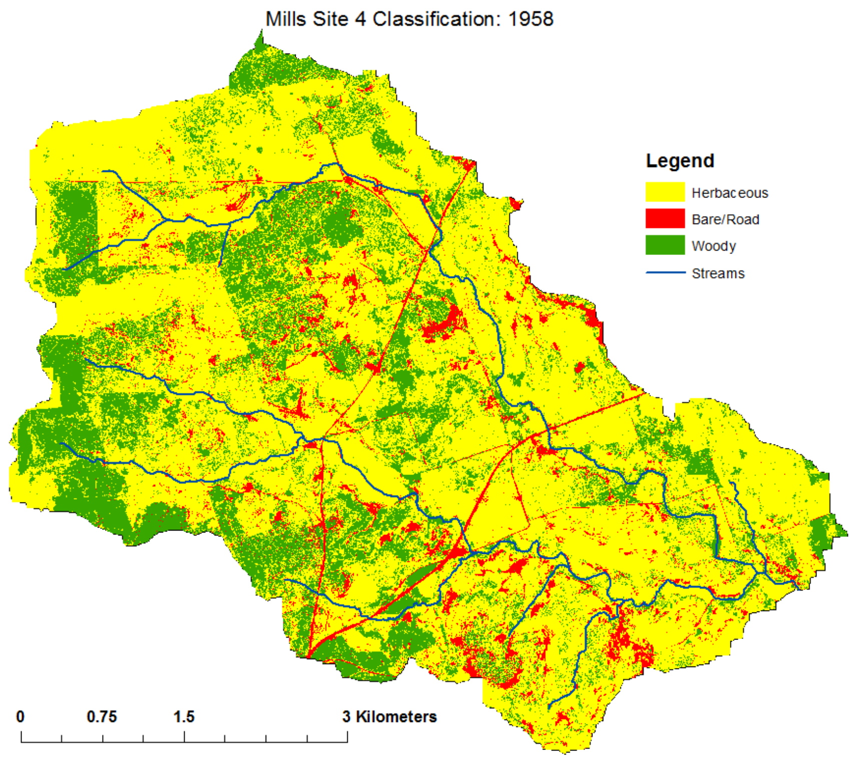


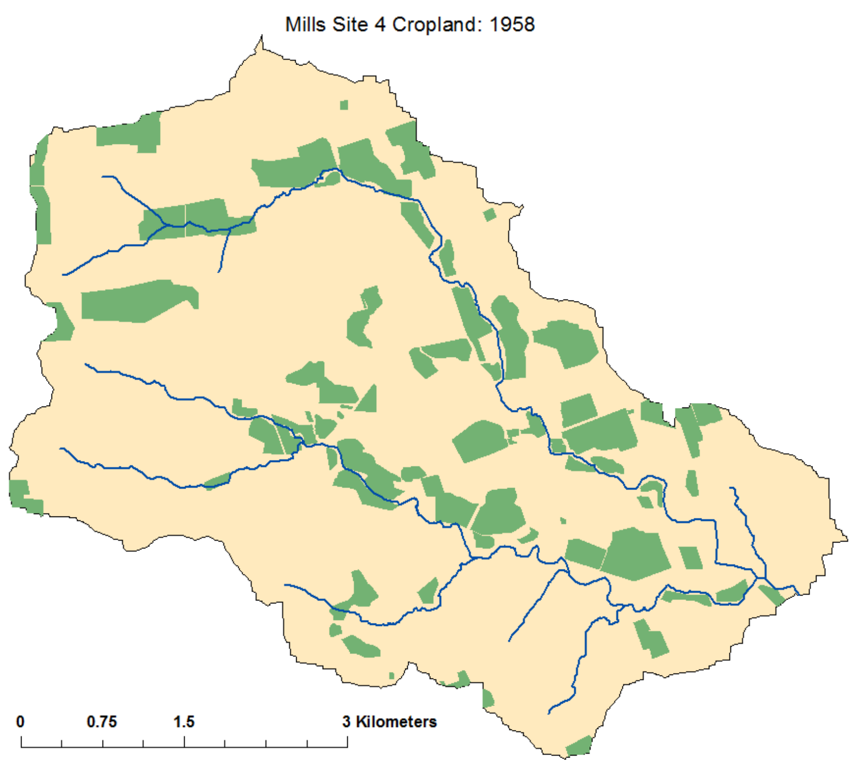
Mills Site 4 Classification: 1937



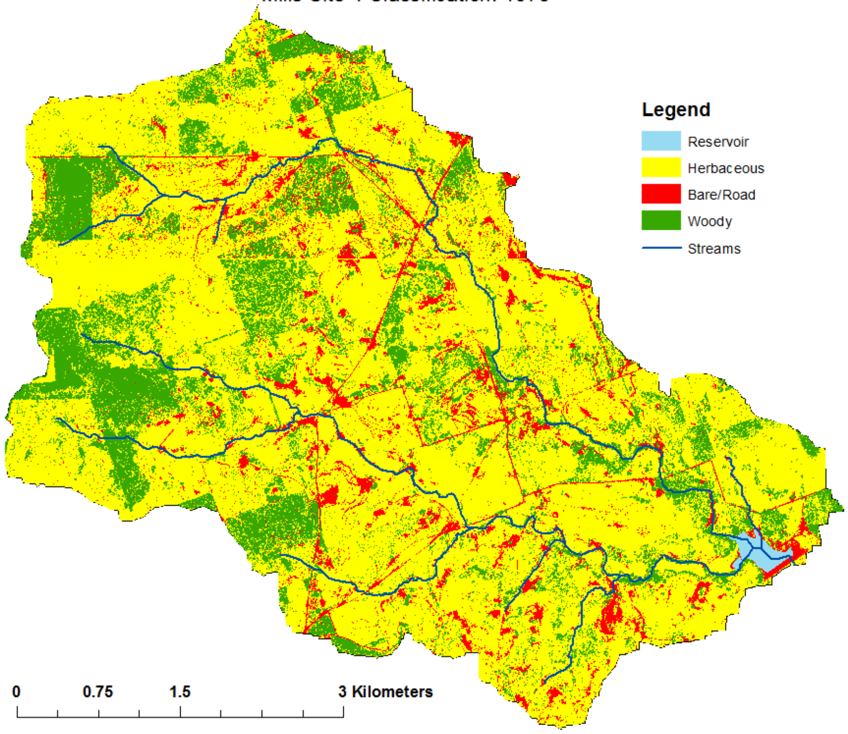
Mills Site 4 Cropland: 1937



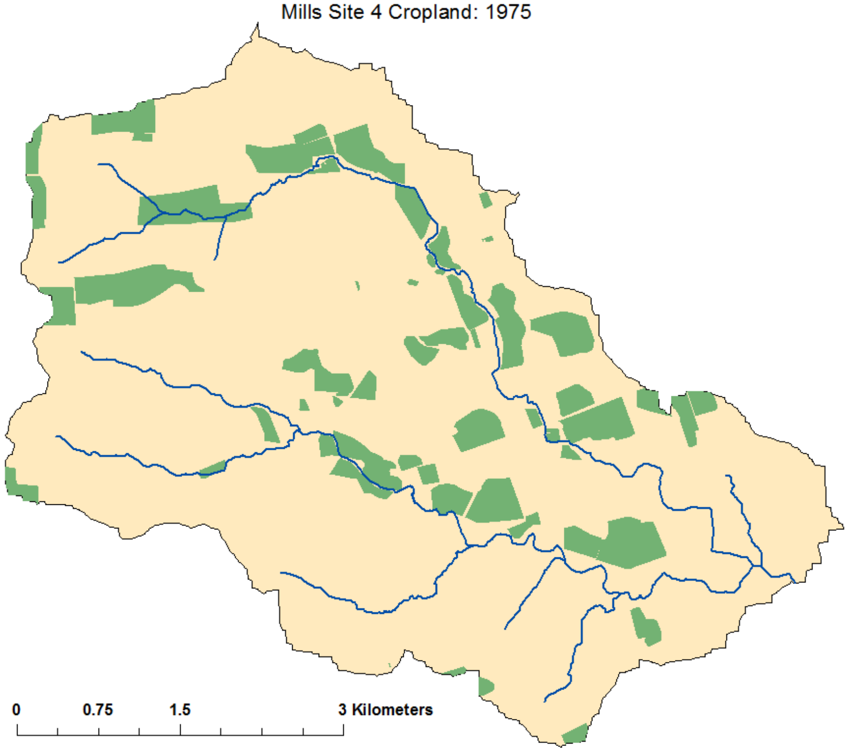

Mills Site 4 Classification: 1958
Legend
Herbaceous
Bare/Road
Woody
Streams
0
0.75
1.5
3 Kilometers


Mills Site 4 Cropland: 1958
0
0.75
1.5
3 Kilometers

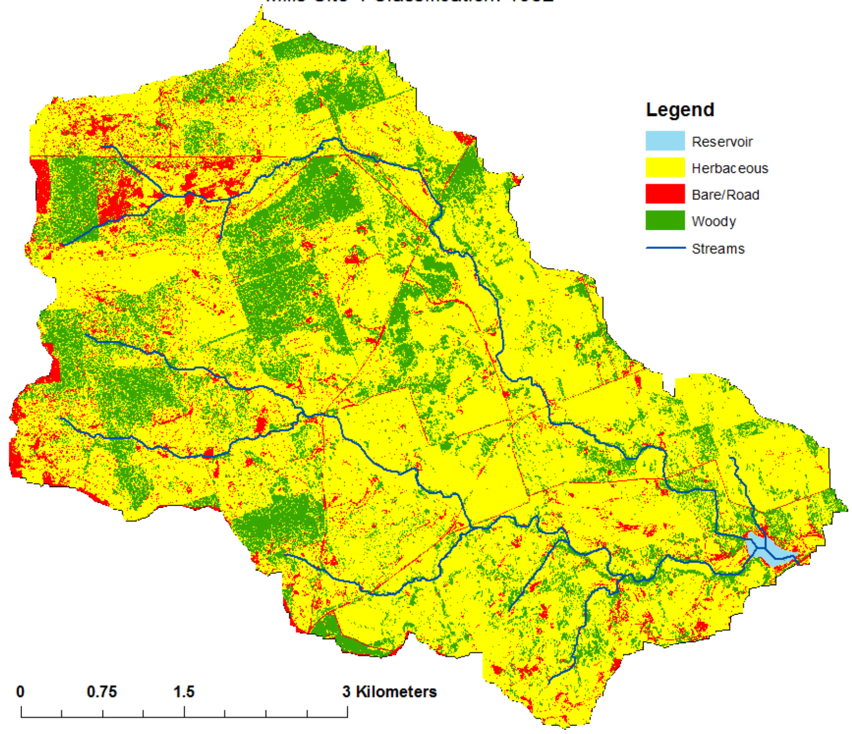
Mills Site 4 Classification: 1975



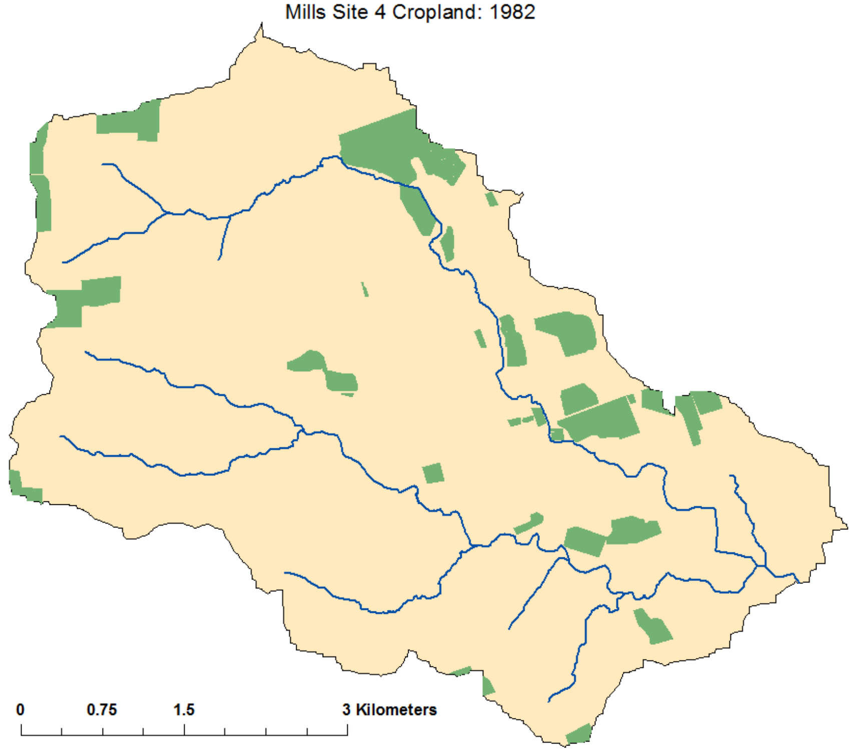
Mills Site 4 Cropland: 1975

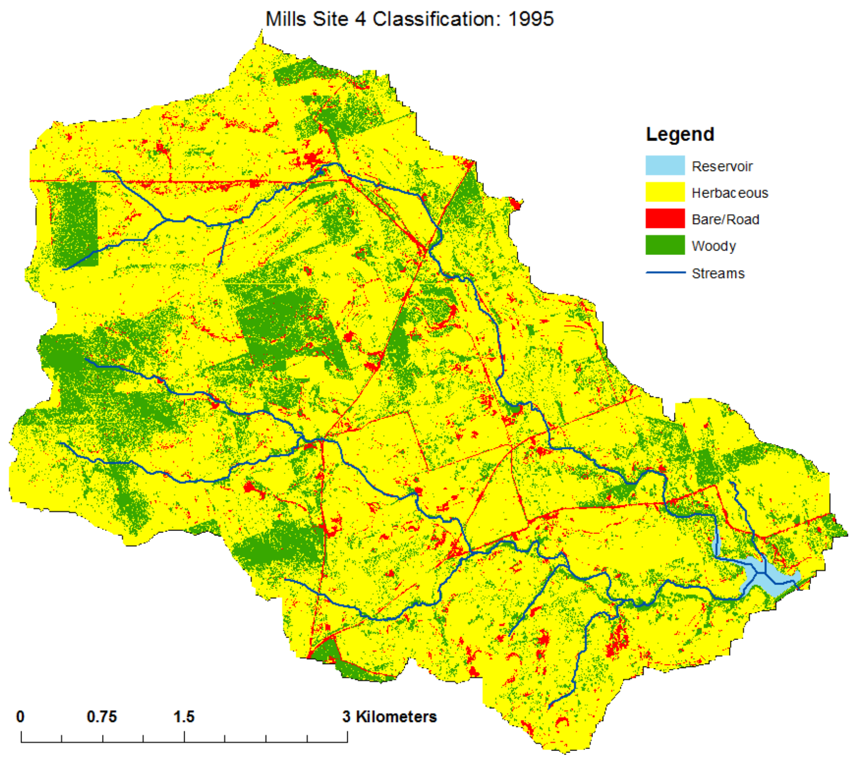


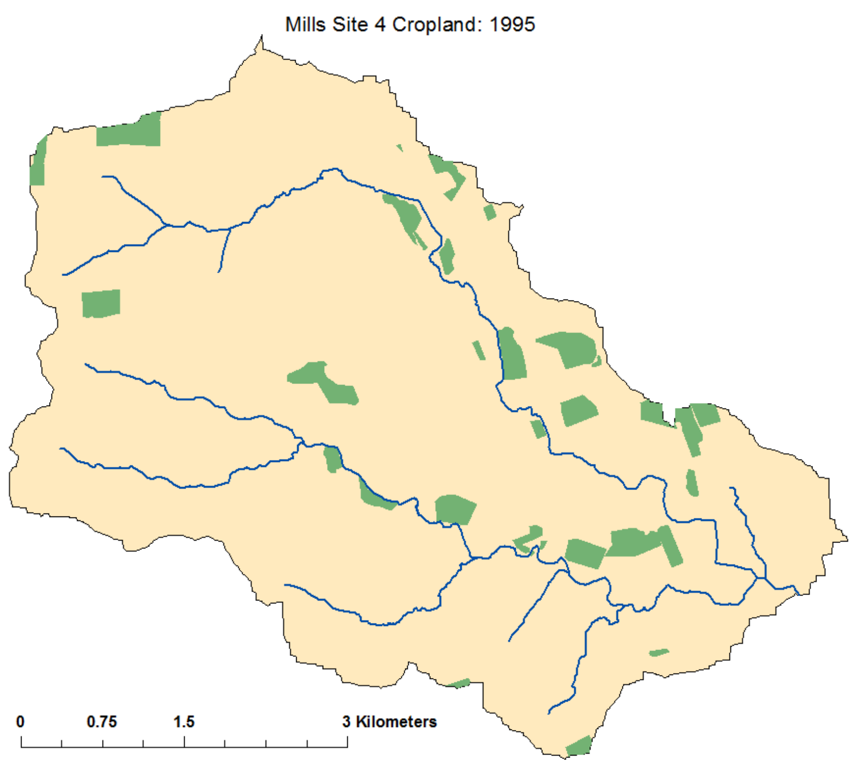
Mills Site 4 Classification: 1982



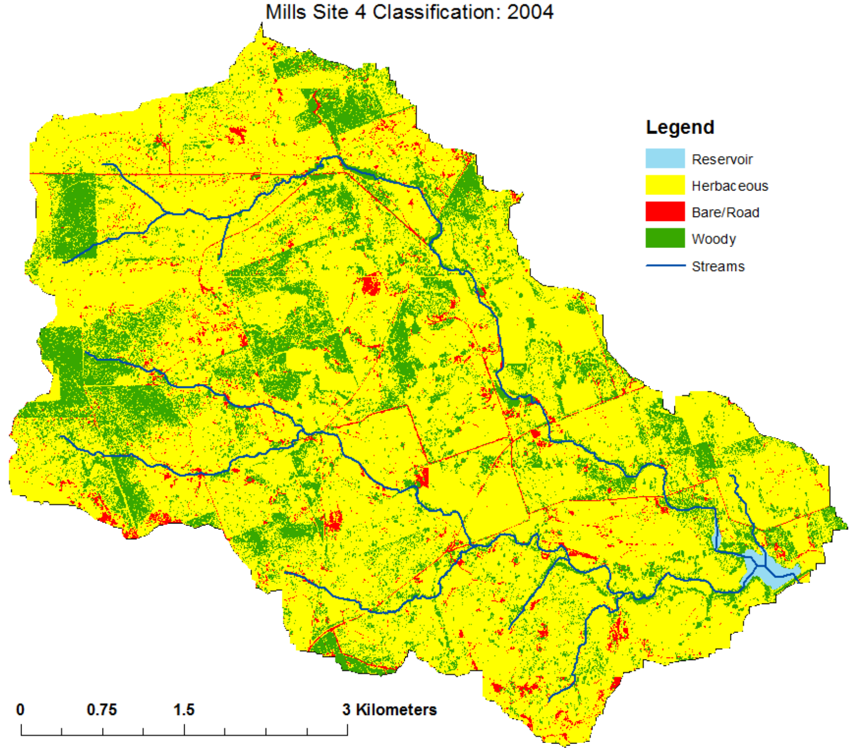
Mills Site 4 Cropland: 1982



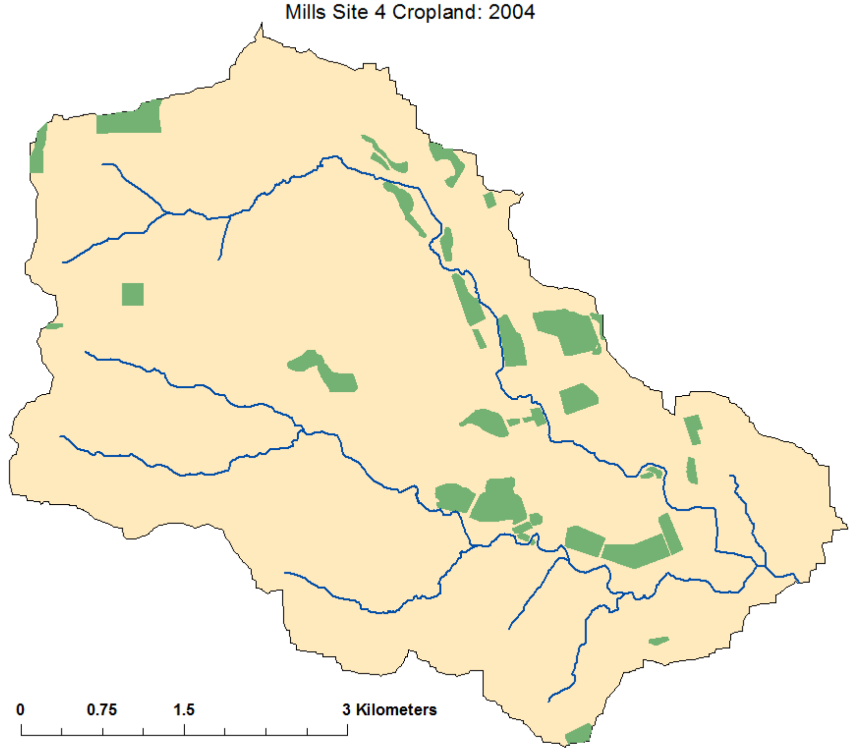

Mills Site 4 Classification: 1995
Legend
Reservoir
Herbaceous
Bare/Road
Woody
Streams
0
0.75
1.5
3 Kilometers


Mills Site 4 Cropland: 1995
0
0.75
1.5
3 Kilometers

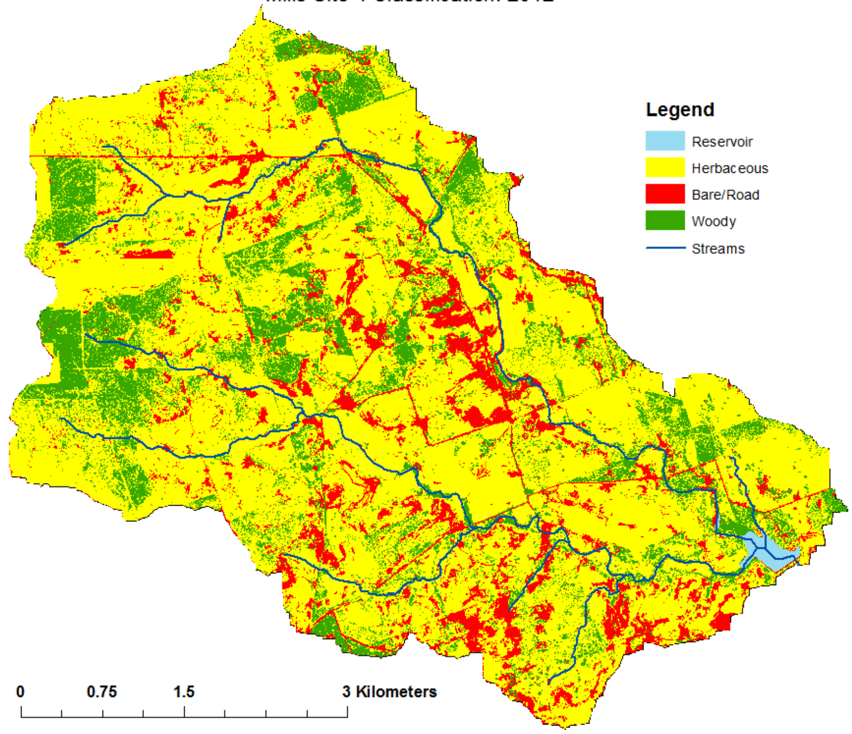
Mills Site 4 Classification: 2004



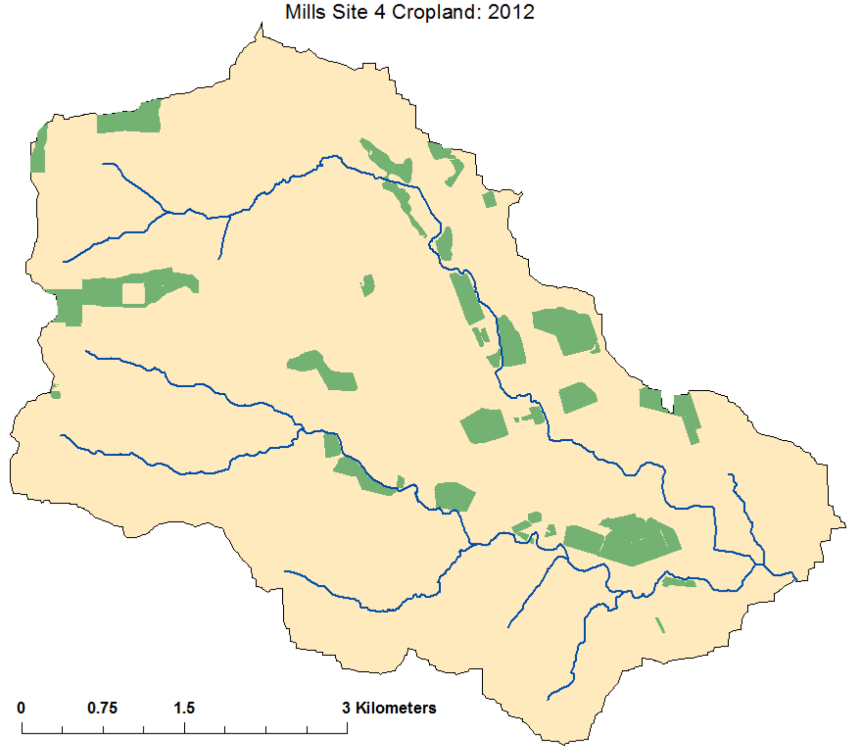
Mills Site 4 Cropland: 2004



Mills Site 4 Classification: 2012

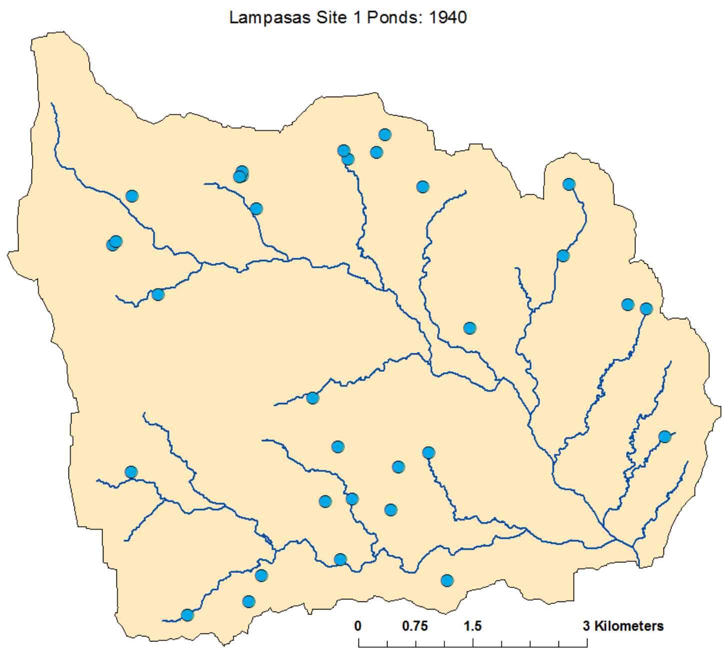


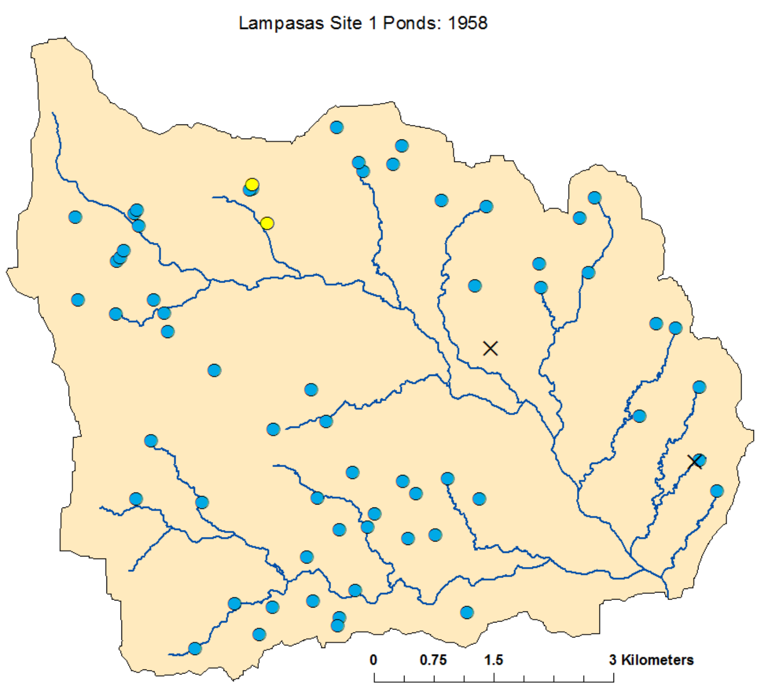
Mills Site 4 Cropland: 2012



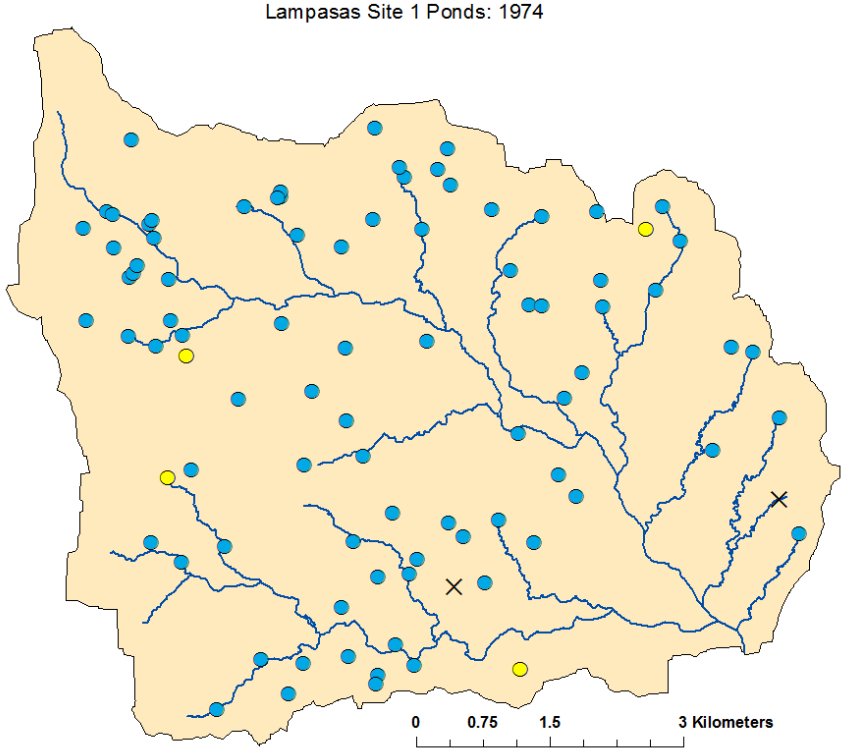
APPENDIX B

WATERSHED POND DISTRIBUTION AND MAINTENANCE BY YEAR

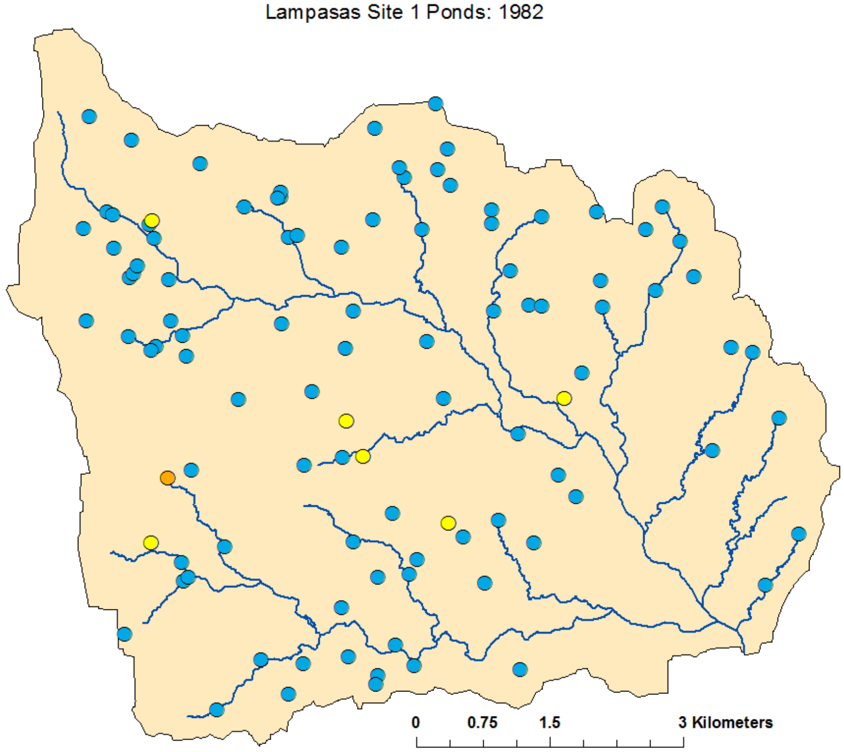




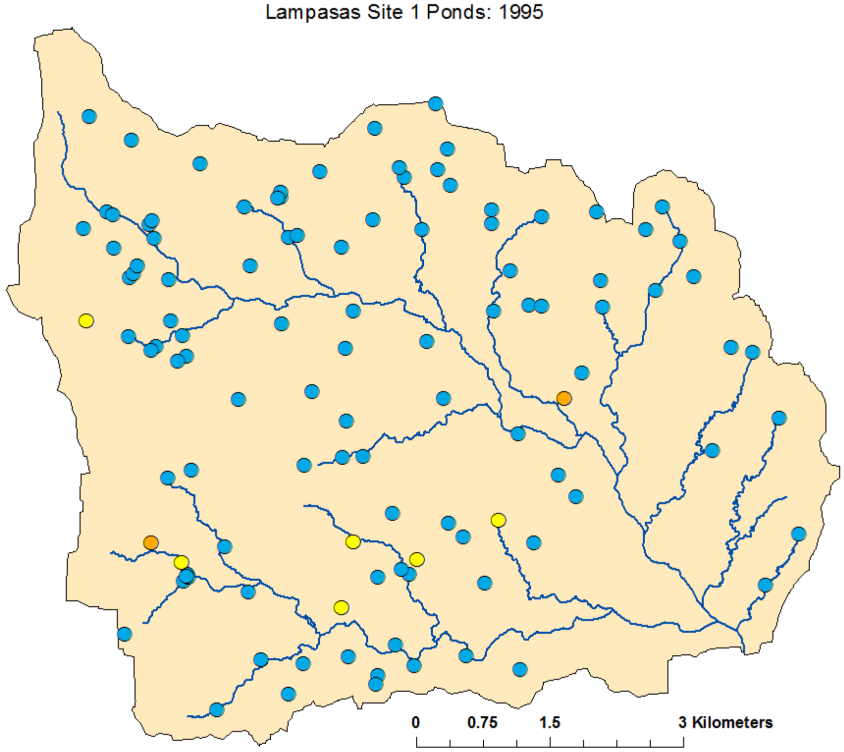
Lampasas Site 1 Ponds: 1974



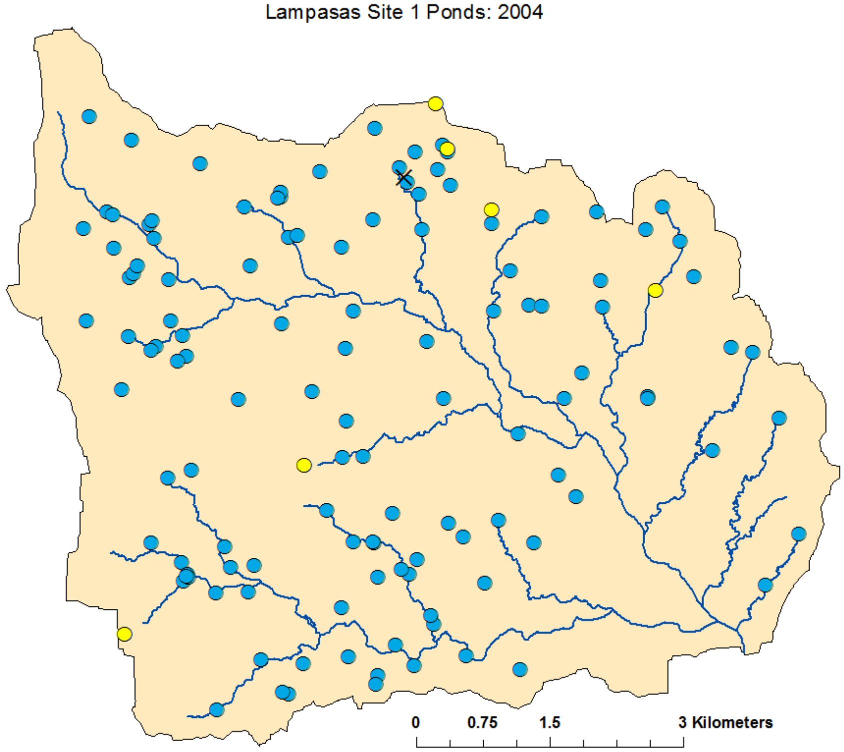
Lampasas Site 1 Ponds: 1982



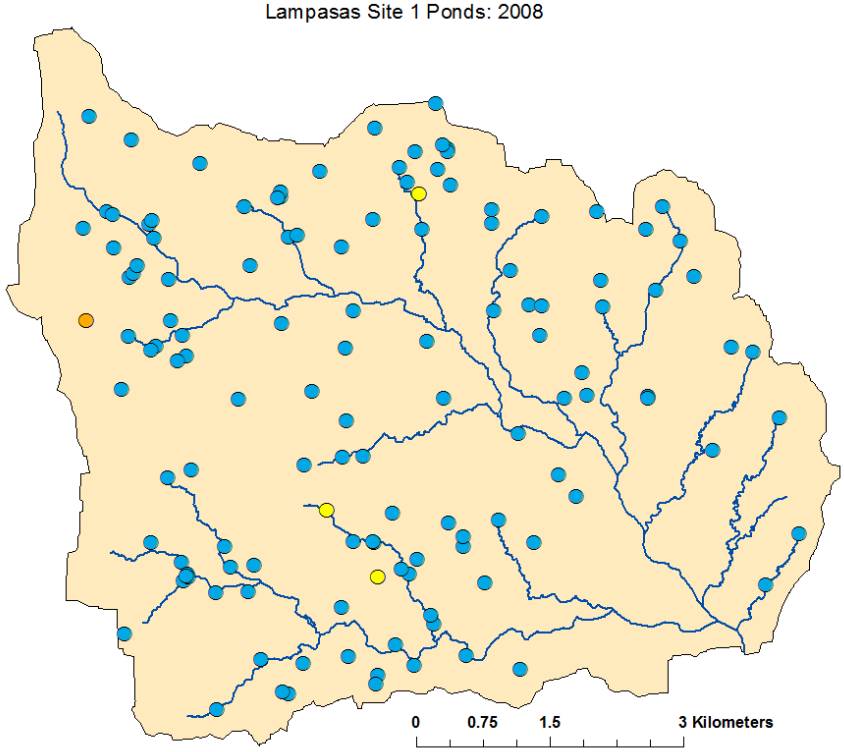
Lampasas Site 1 Ponds: 1995



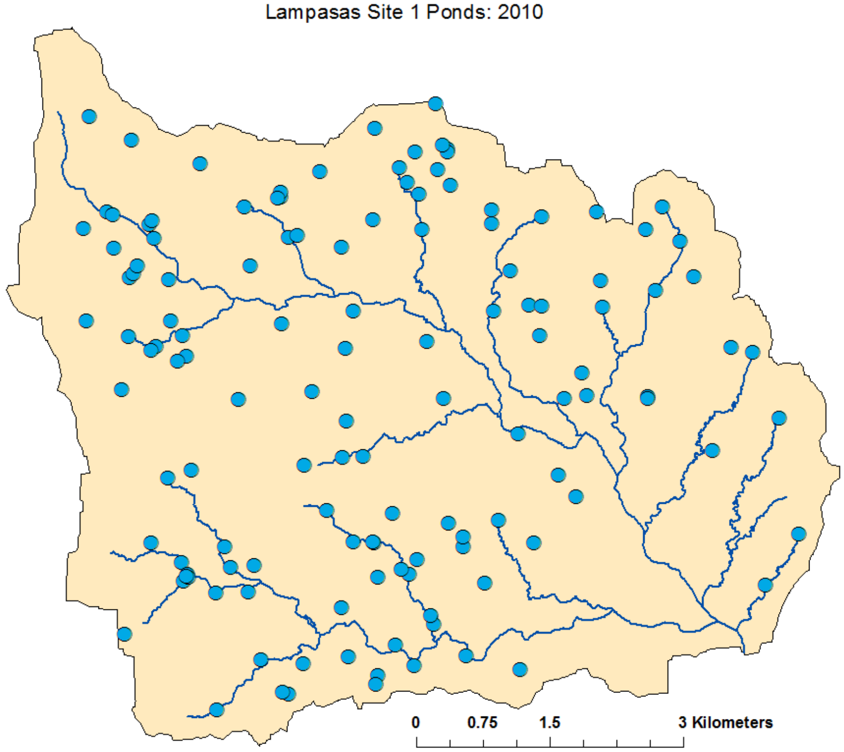
Lampasas Site 1 Ponds: 2004



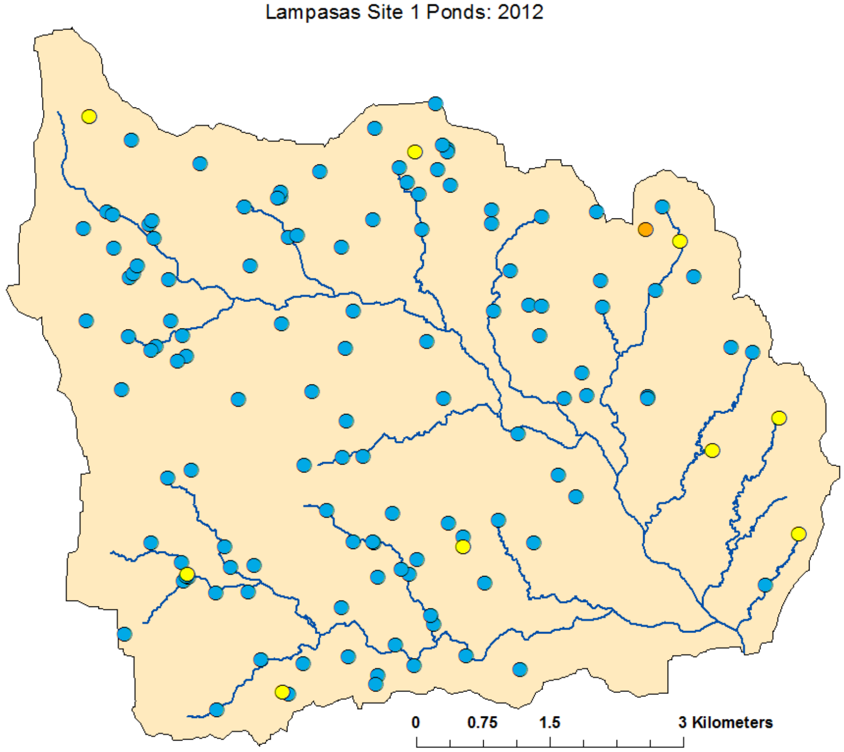
Lampasas Site 1 Ponds: 2008



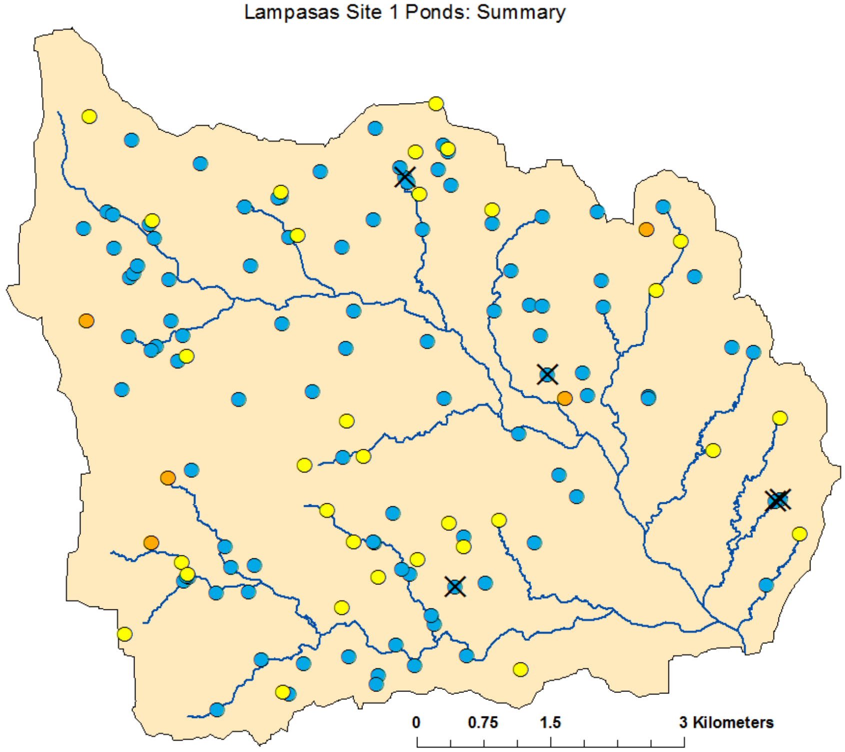
Lampasas Site 1 Ponds: 2010



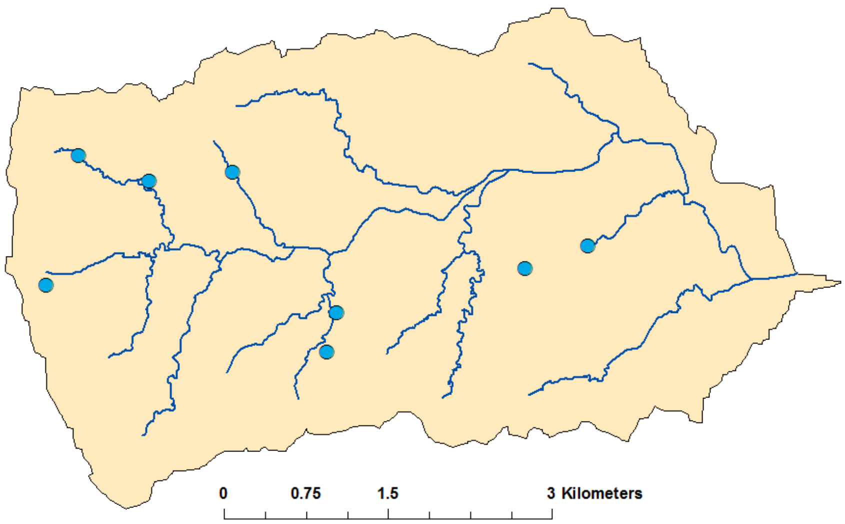
Lampasas Site 1 Ponds: 2012



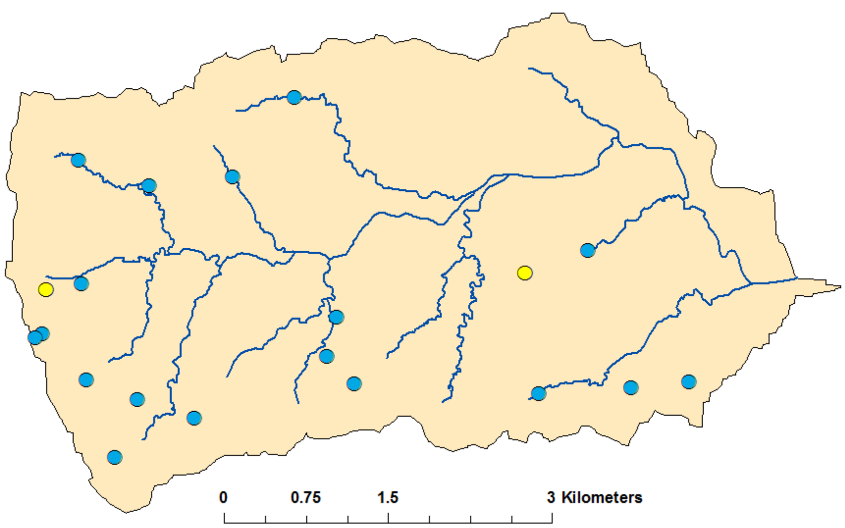
Lampasas Site 1 Ponds: Summary



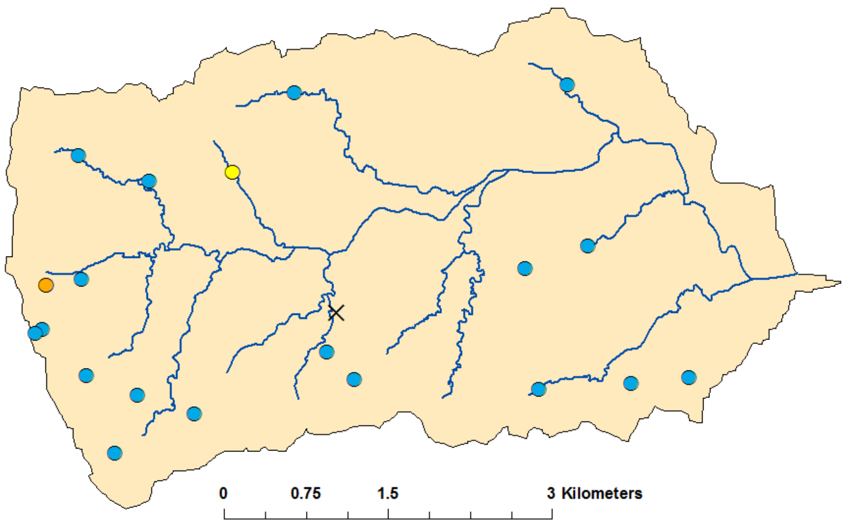
Lampasas Site 2 Ponds: 1940



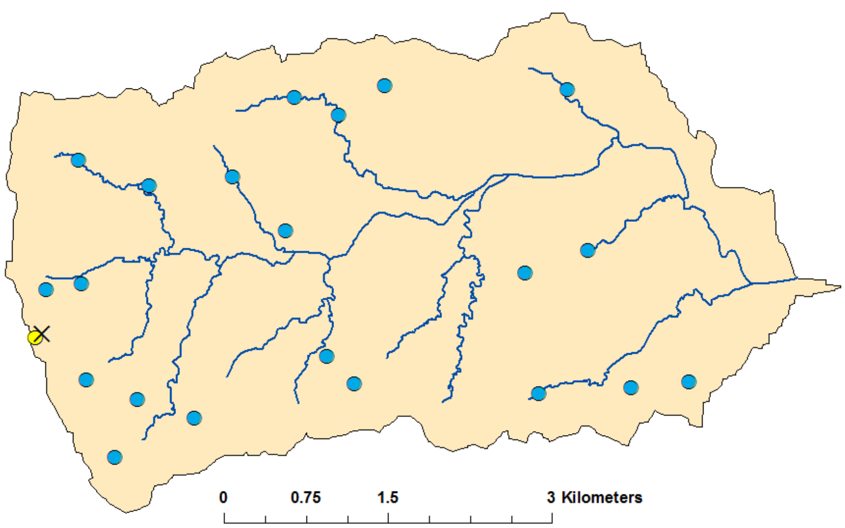
Lampasas Site 2 Ponds: 1958



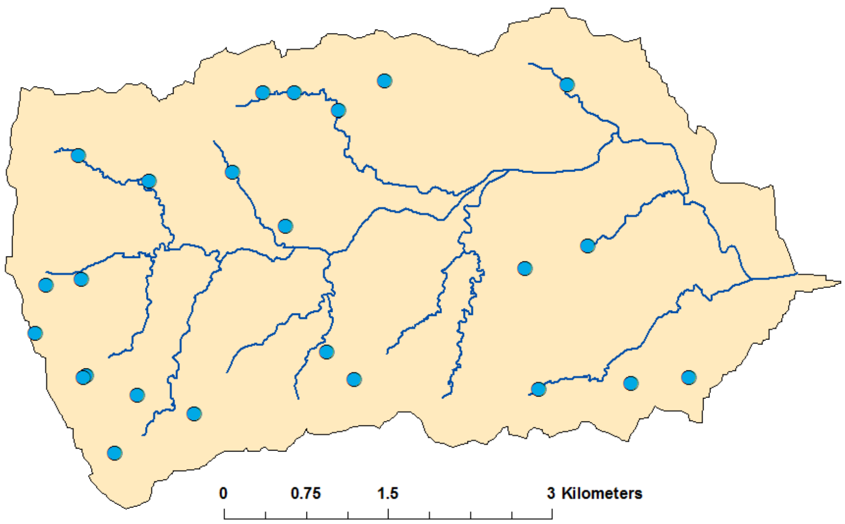
Lampasas Site 2 Ponds: 1974



Lampasas Site 2 Ponds: 1982



Lampasas Site 2 Ponds: 1995



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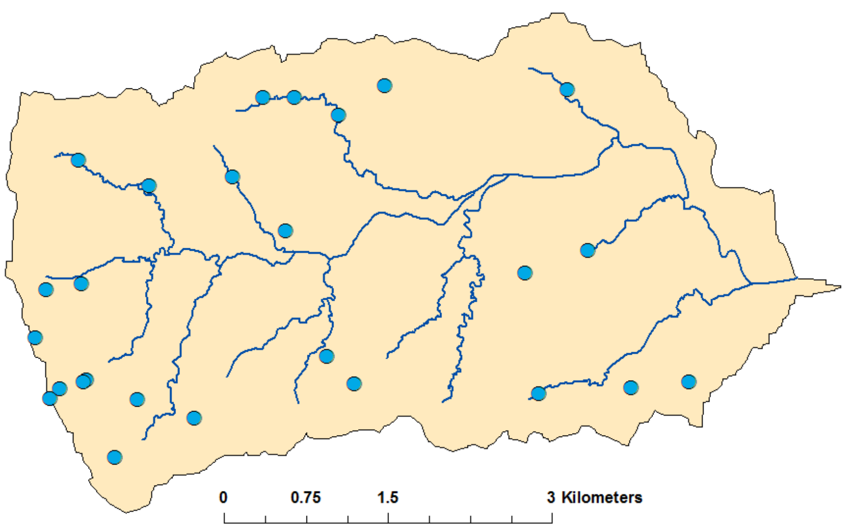

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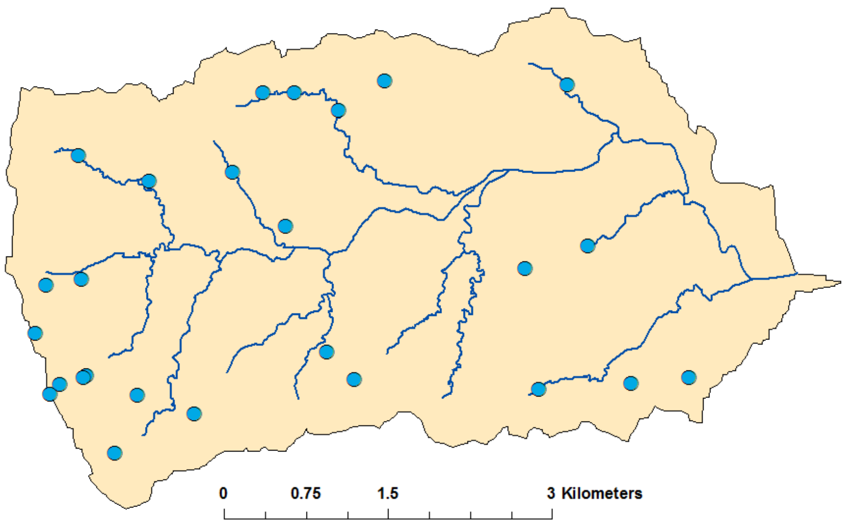
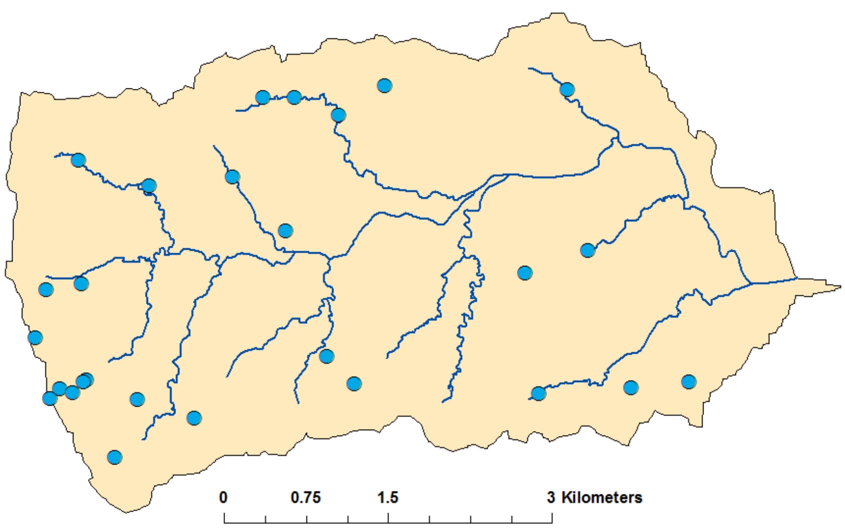
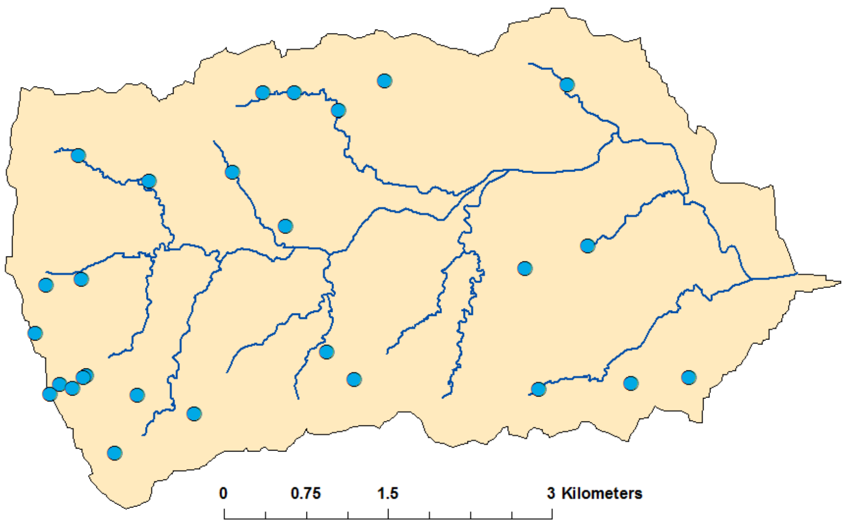


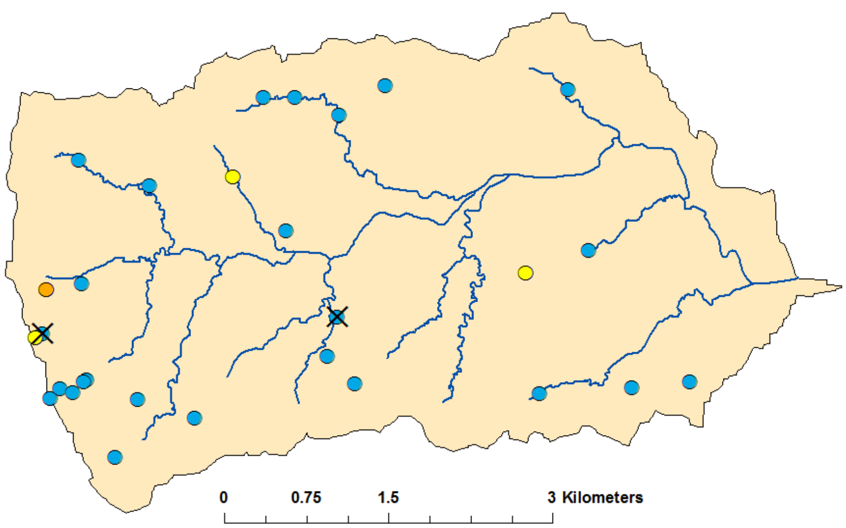
Lampasas Site 2 Ponds: 2010



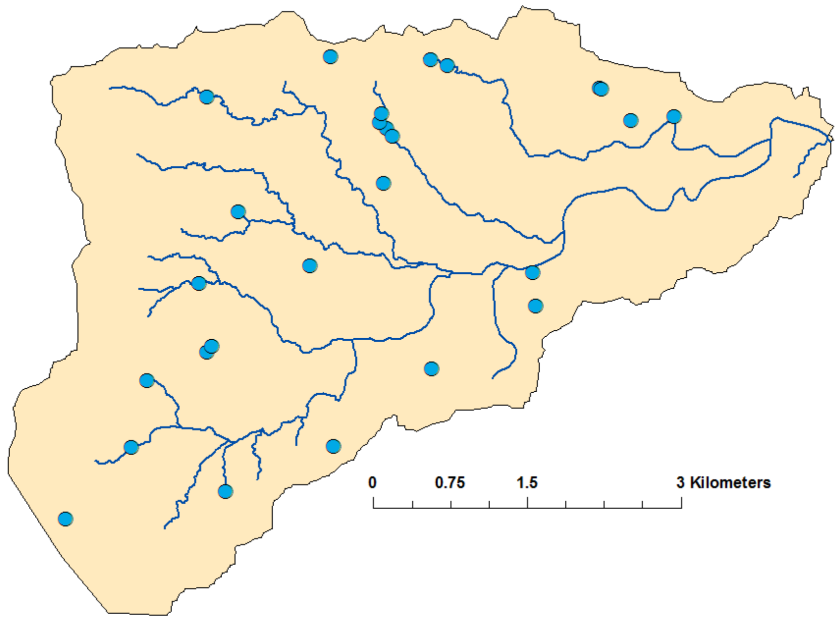
Lampasas Site 2 Ponds: 2012



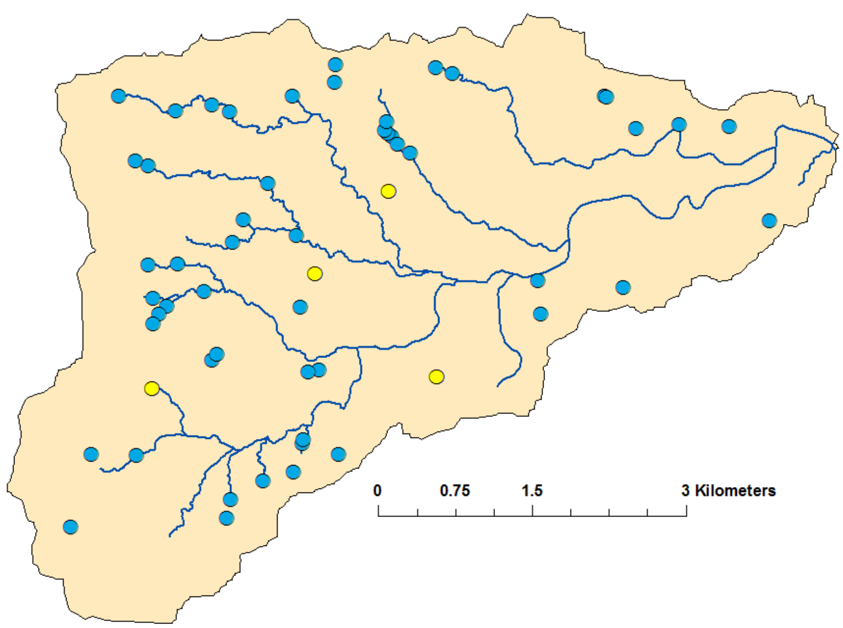
Lampasas Site 2 Ponds: Summary



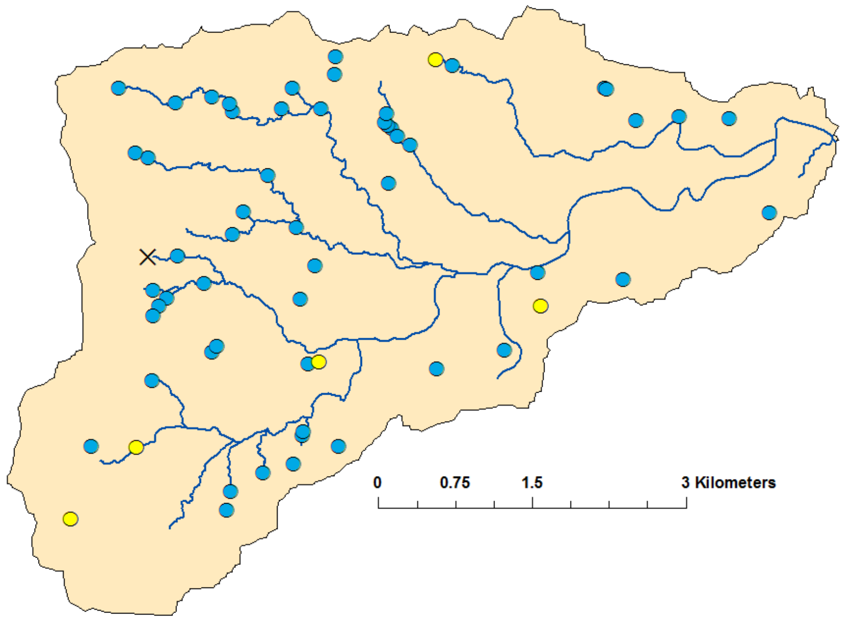
Lampasas Site 3 Ponds: 1940



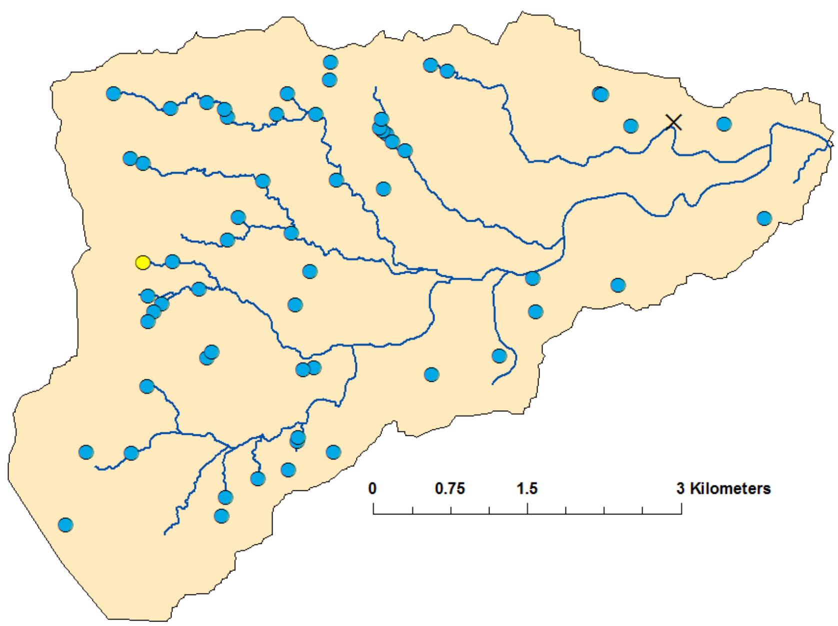
Lampasas Site 3 Ponds: 1958



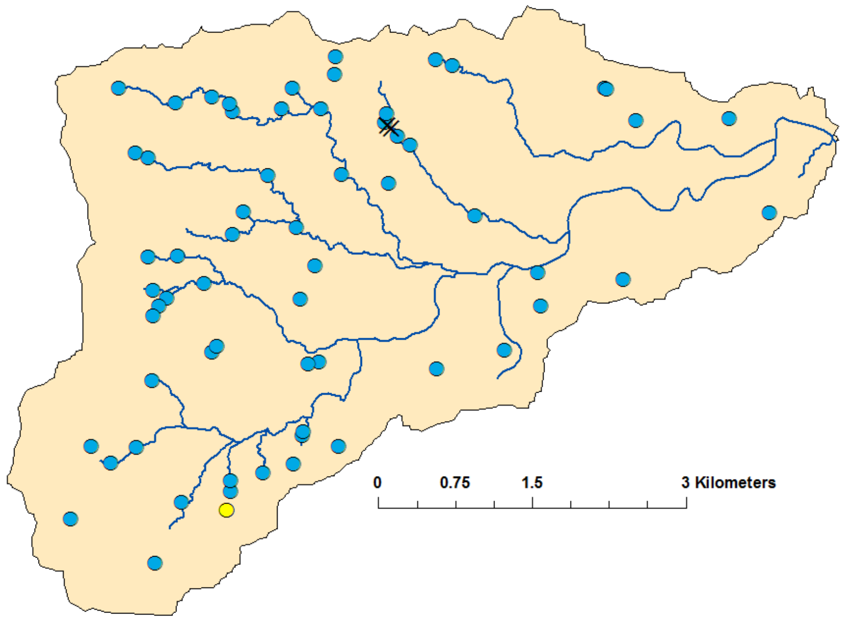
Lampasas Site 3 Ponds: 1974



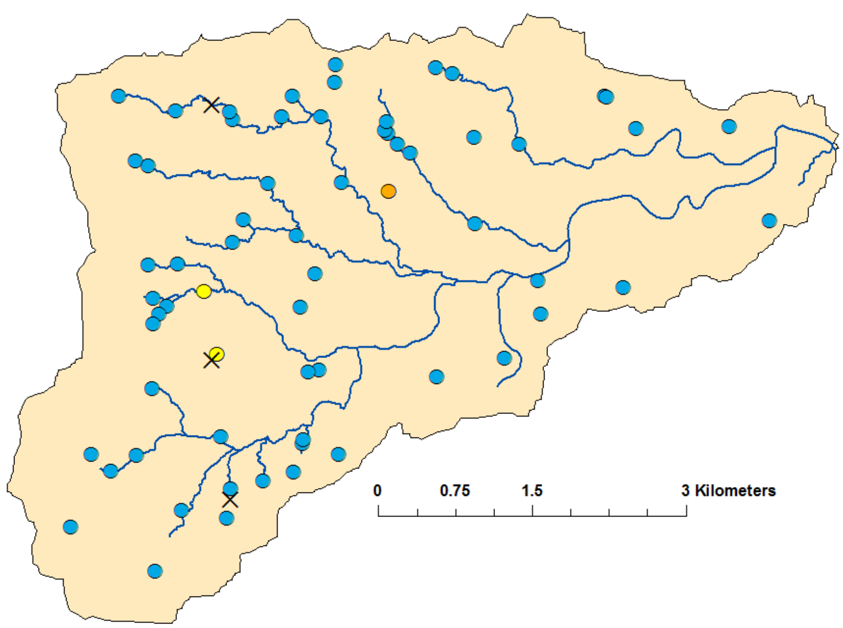
Lampasas Site 3 Ponds: 1982



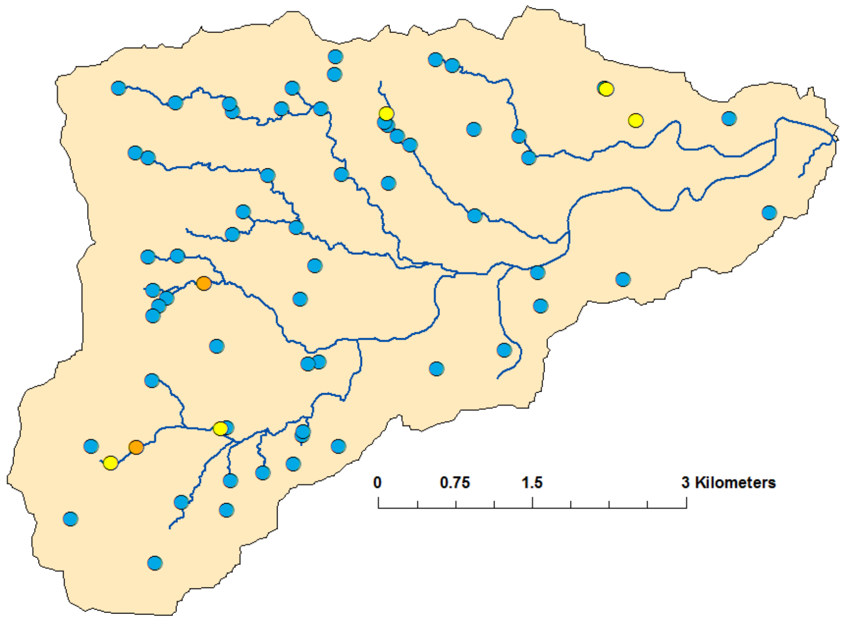
Lampasas Site 3 Ponds: 1995



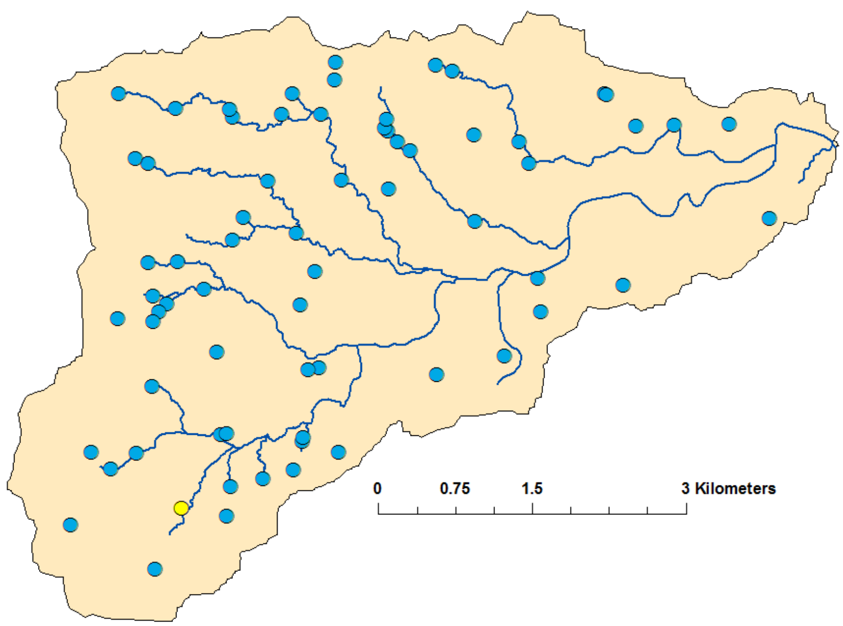
Lampasas Site 3 Ponds: 2004



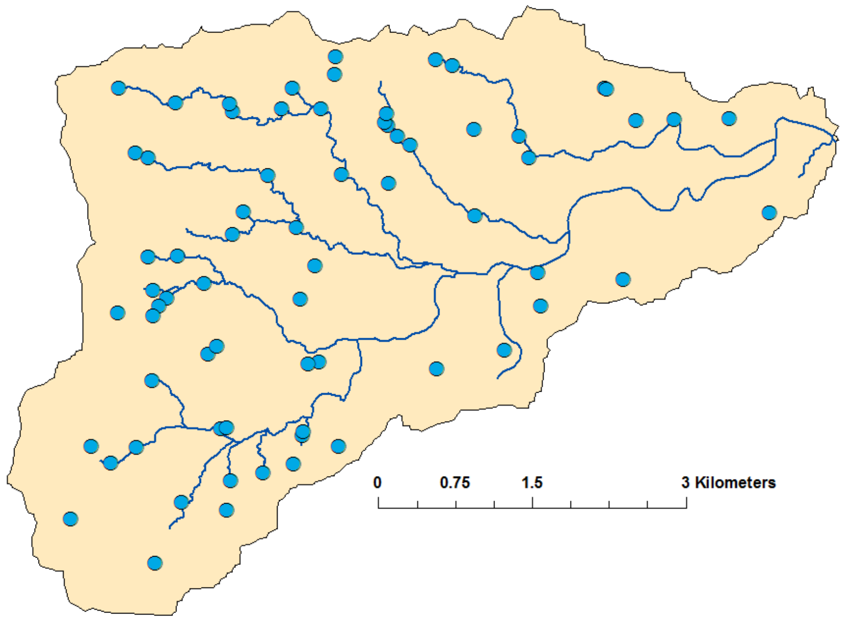
Lampasas Site 3 Ponds: 2008



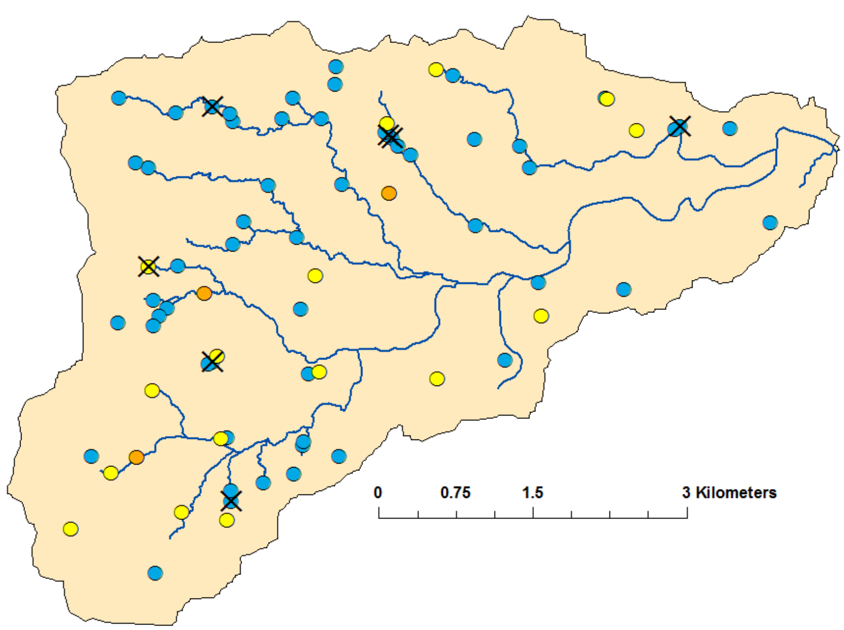
Lampasas Site 3 Ponds: 2010



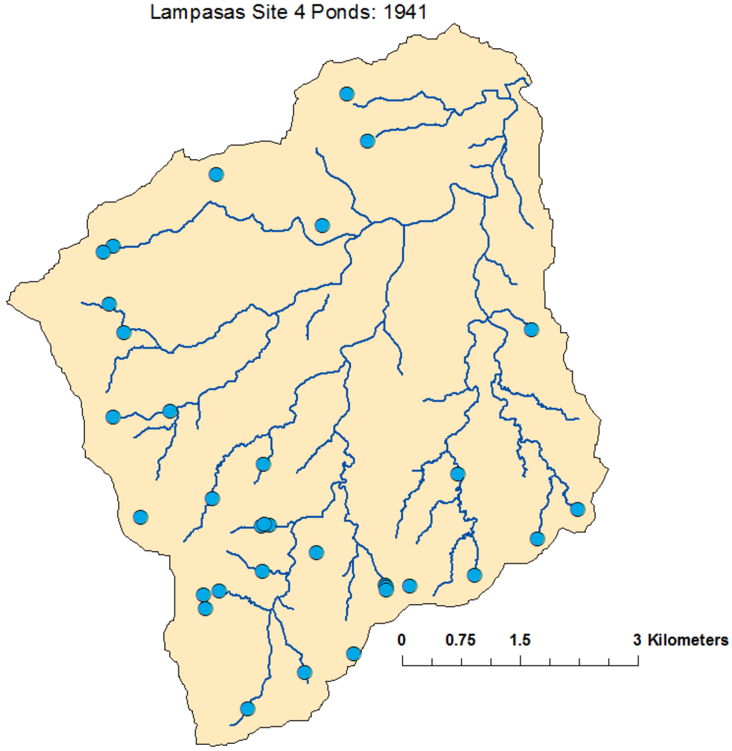
Lampasas Site 3 Ponds: 2012



Lampasas Site 3 Ponds: Summary



Lampasas Site 4 Ponds: 1941



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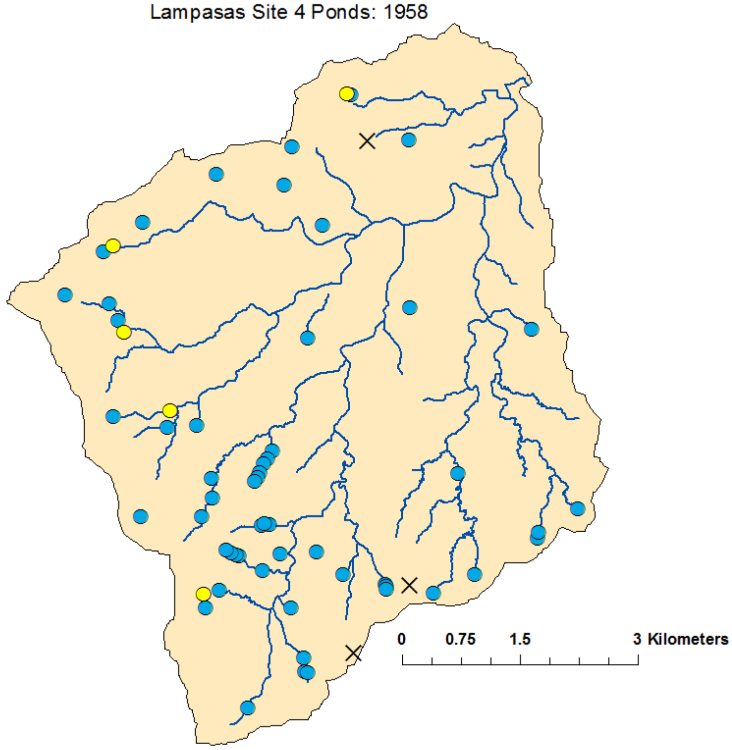
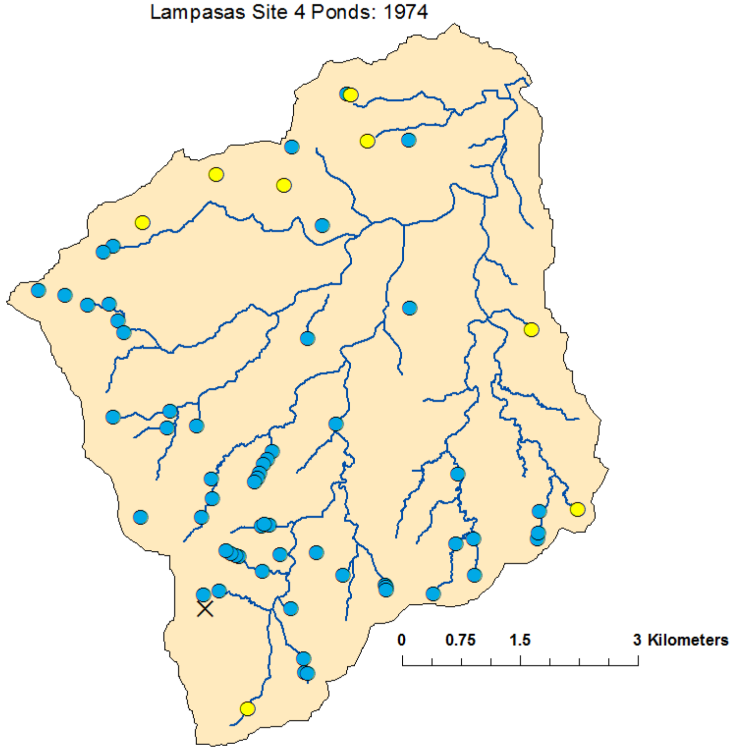
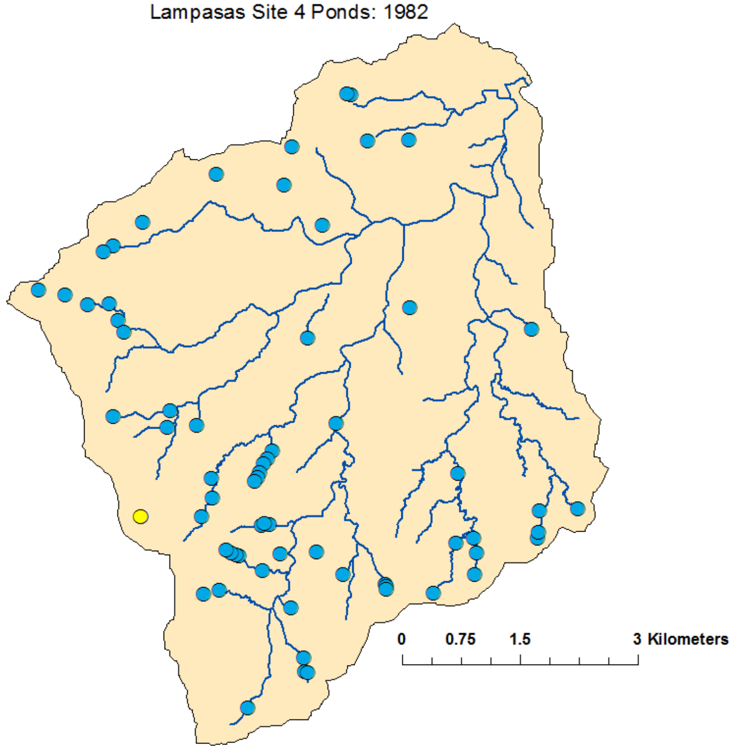
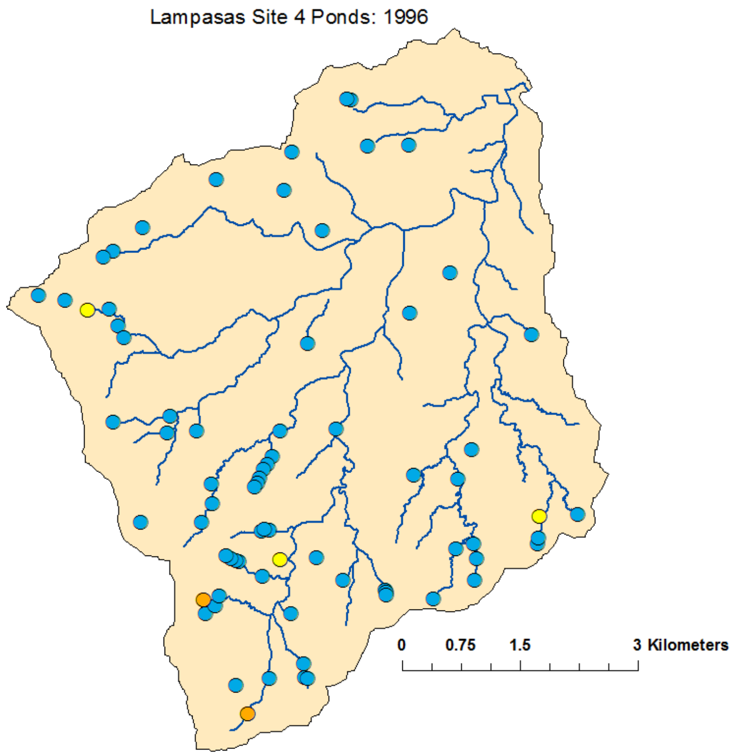


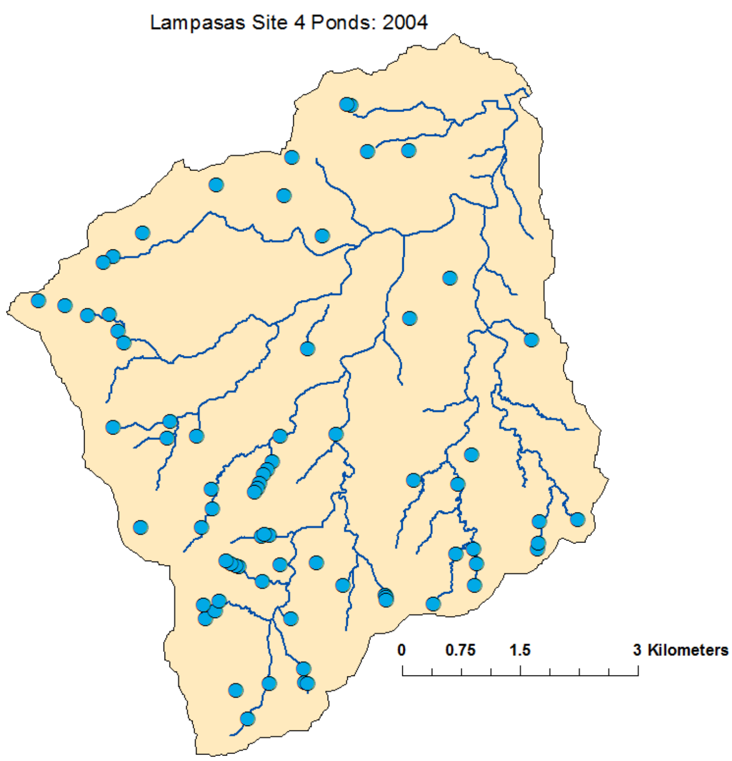
Lampasas Site 4 Ponds: 1974



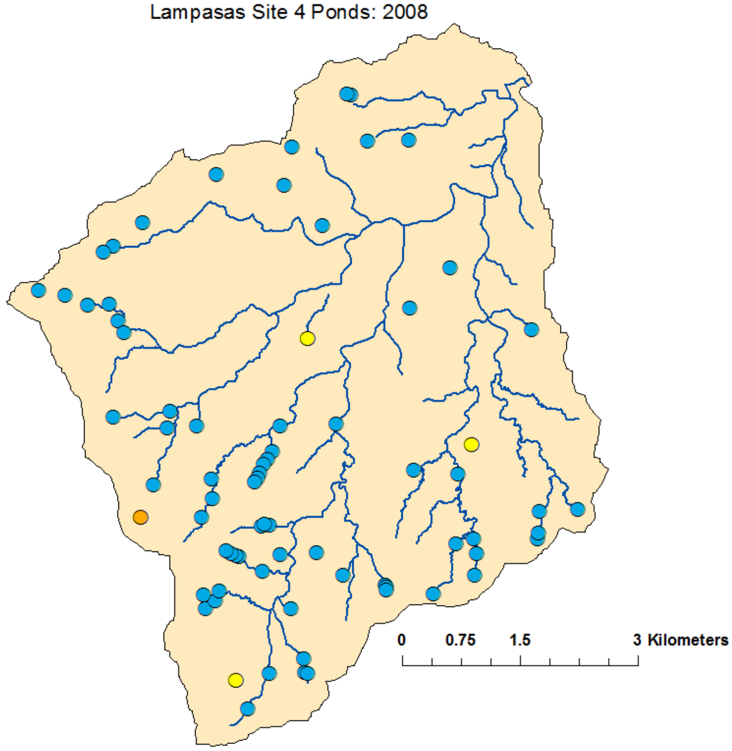
Lampasas Site 4 Ponds: 1982



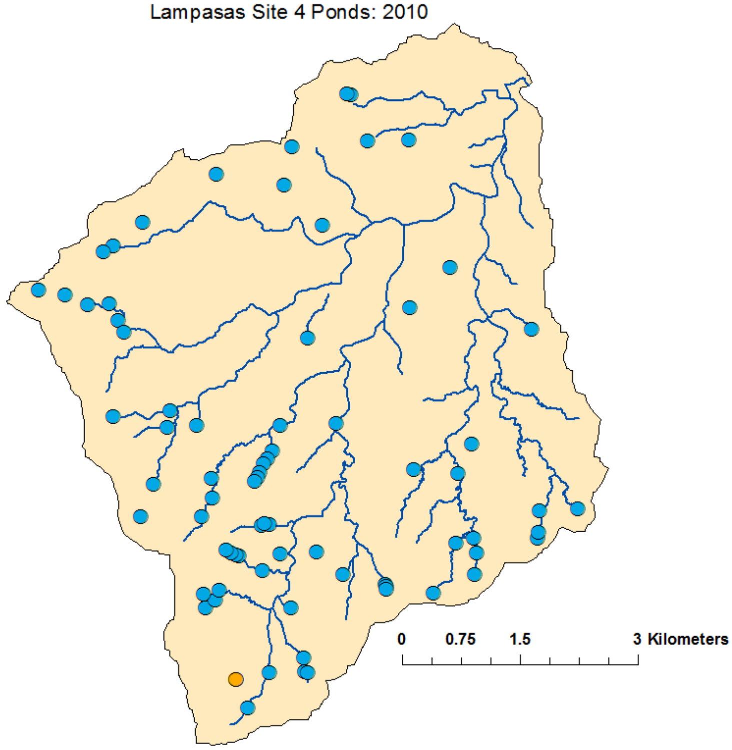

Lampasas Site 4 Ponds: 1996
0
0.75
1.5
3 Kilometers


Lampasas Site 4 Ponds: 2004
0
0.75
1.5
3 Kilometers

Lampasas Site 4 Ponds: 2008



Lampasas Site 4 Ponds: 2010



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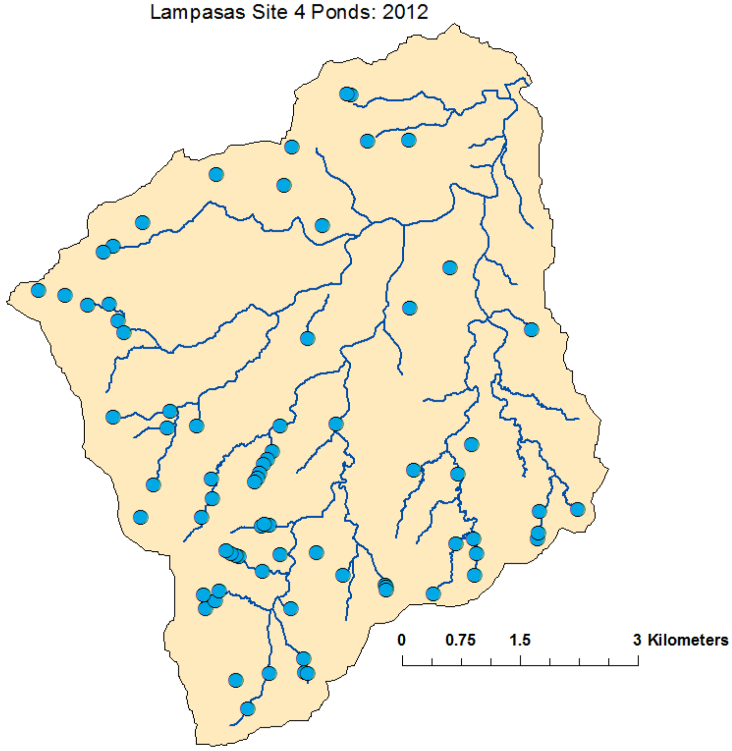
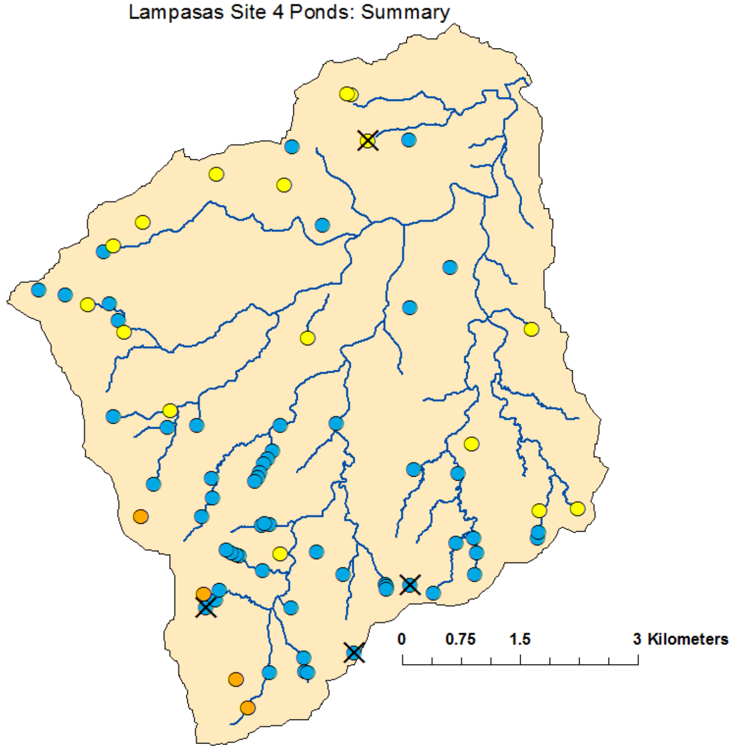
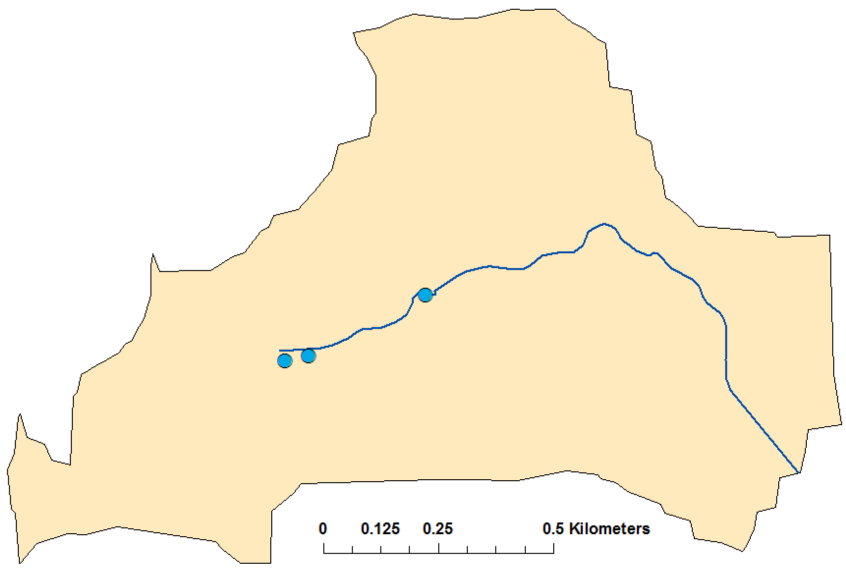


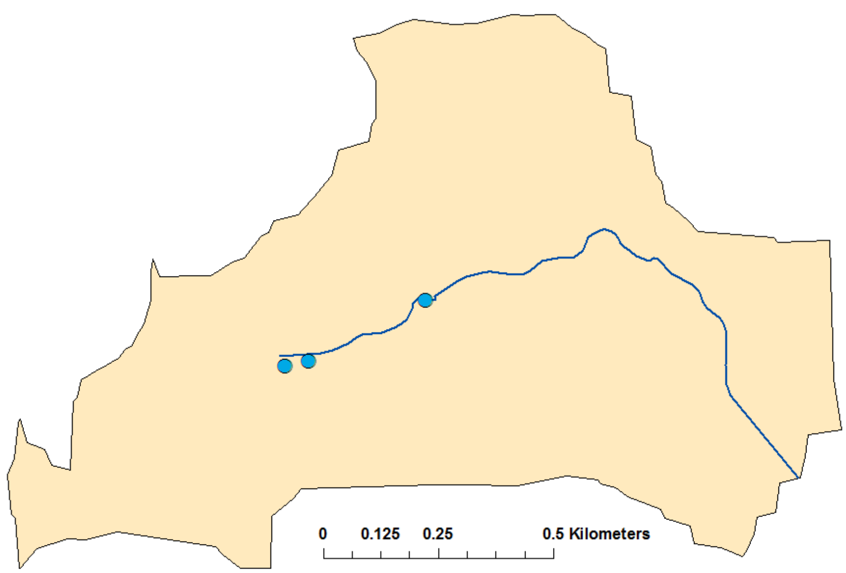
Lampasas Site 4 Ponds: Summary



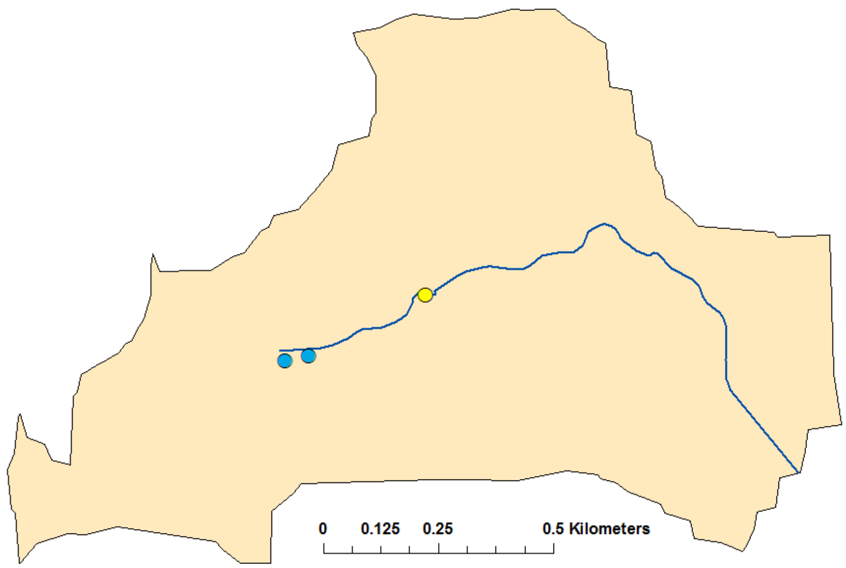
Lampasas Site 9 Ponds: 1941



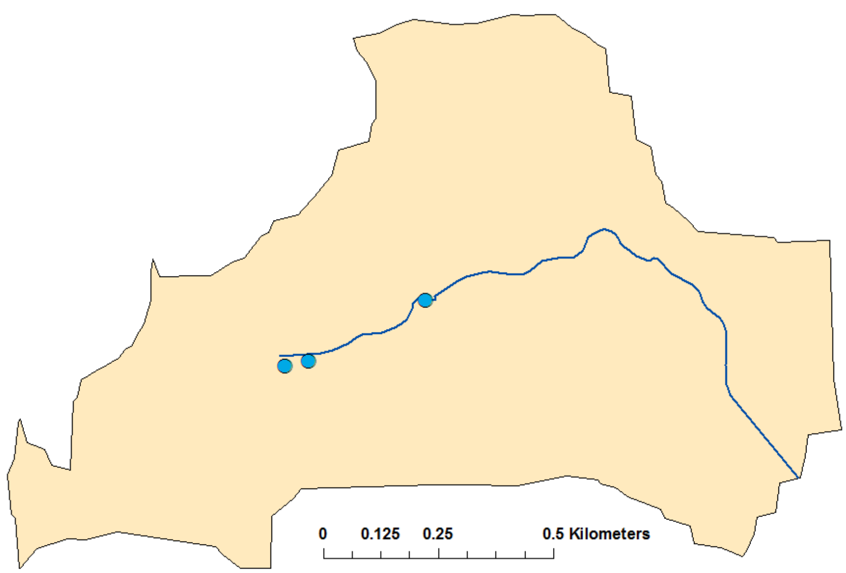
Lampasas Site 9 Ponds: 1958



Lampasas Site 9 Ponds: 1974



Lampasas Site 9 Ponds: 1982



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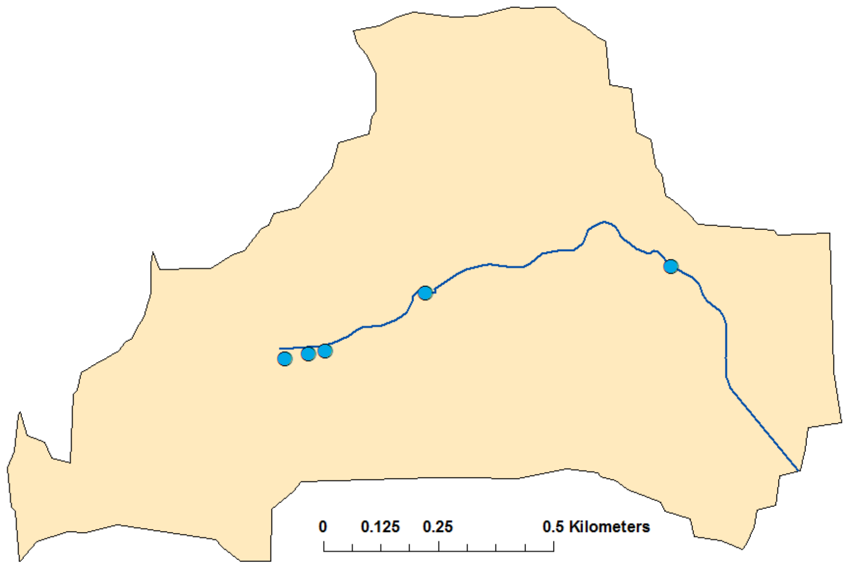

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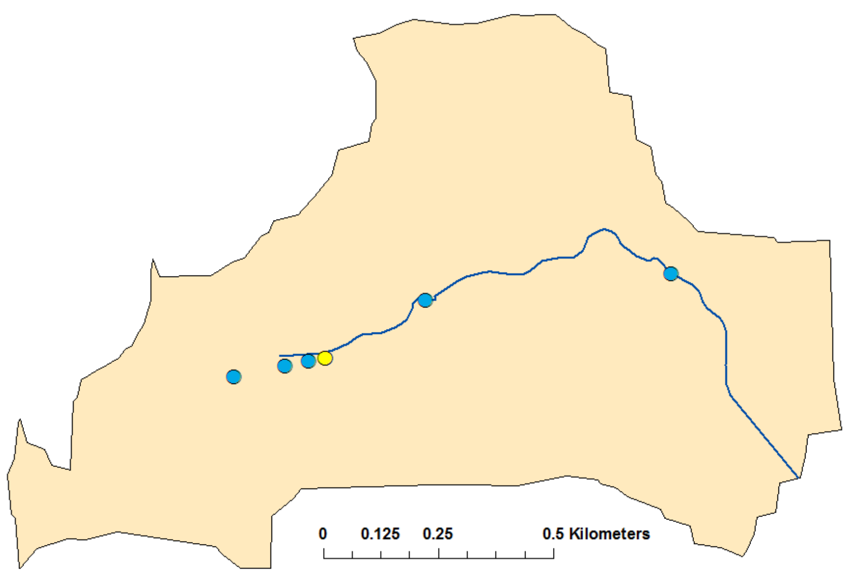

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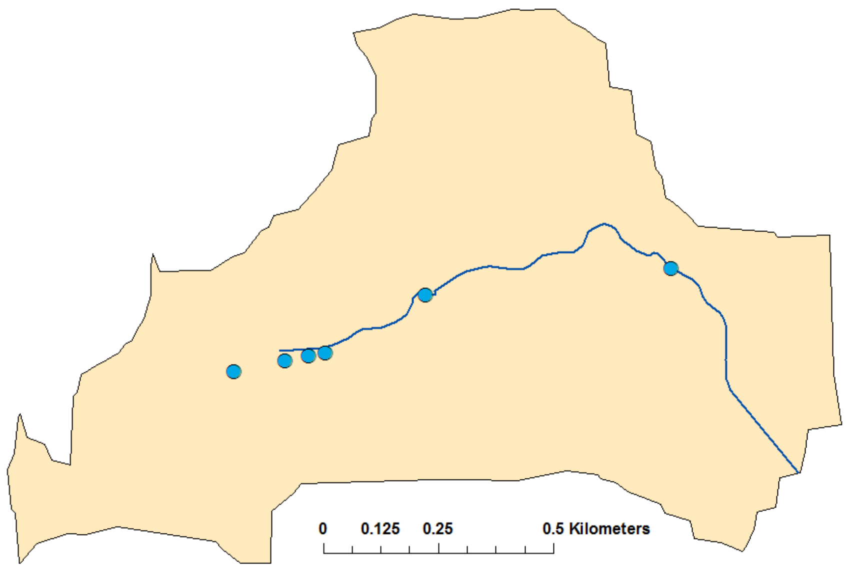
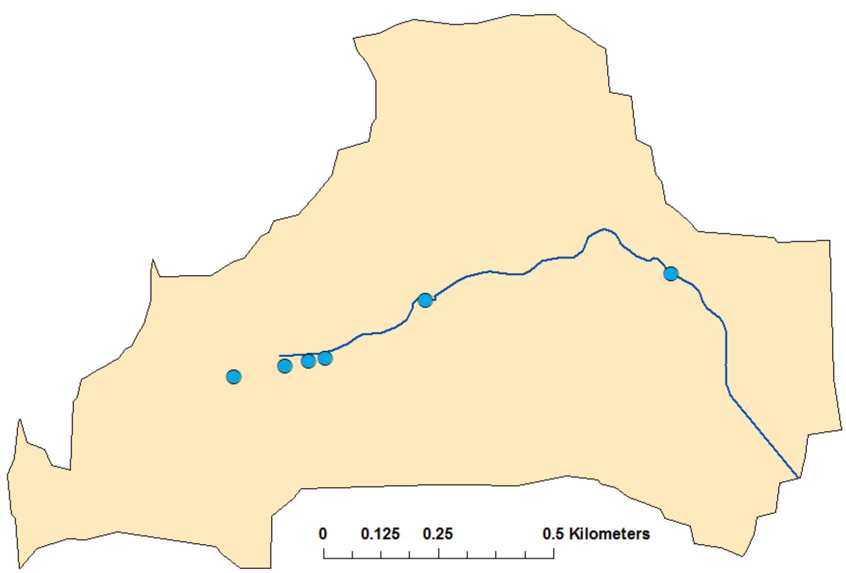
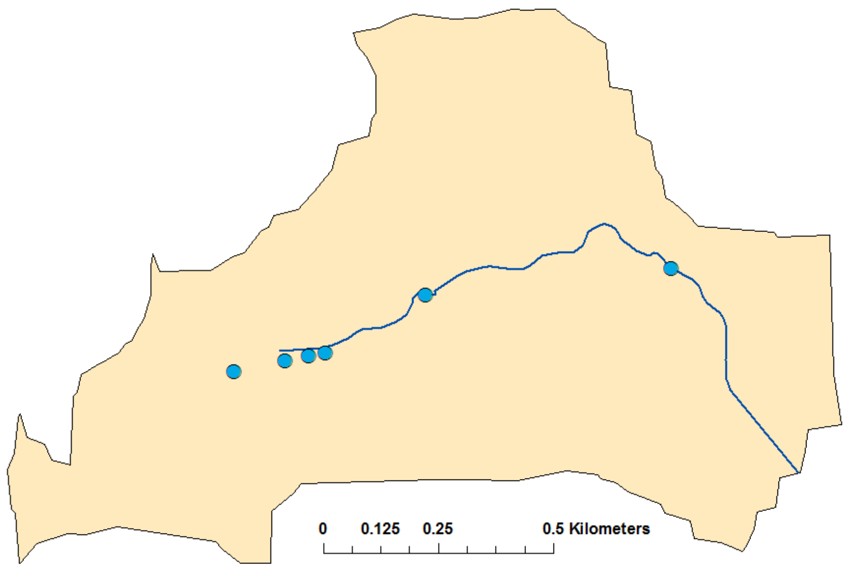


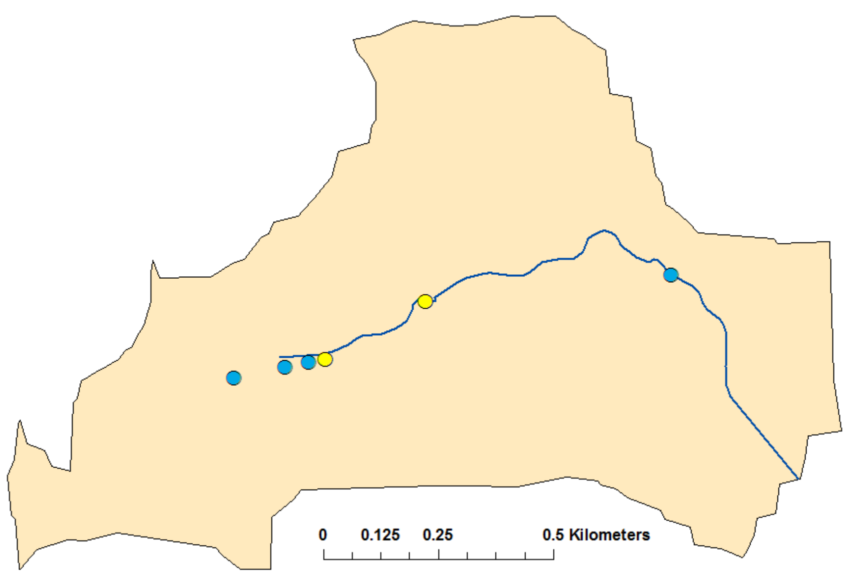
Lampasas Site 9 Ponds: 2010



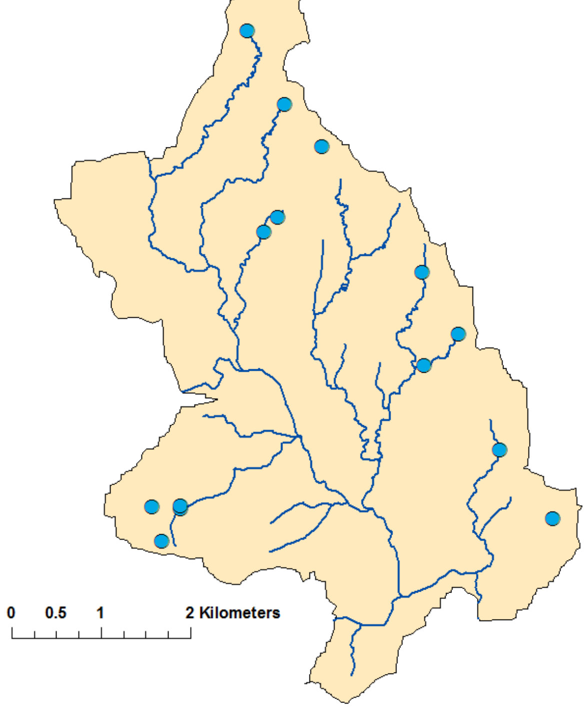
Lampasas Site 9 Ponds: 2012



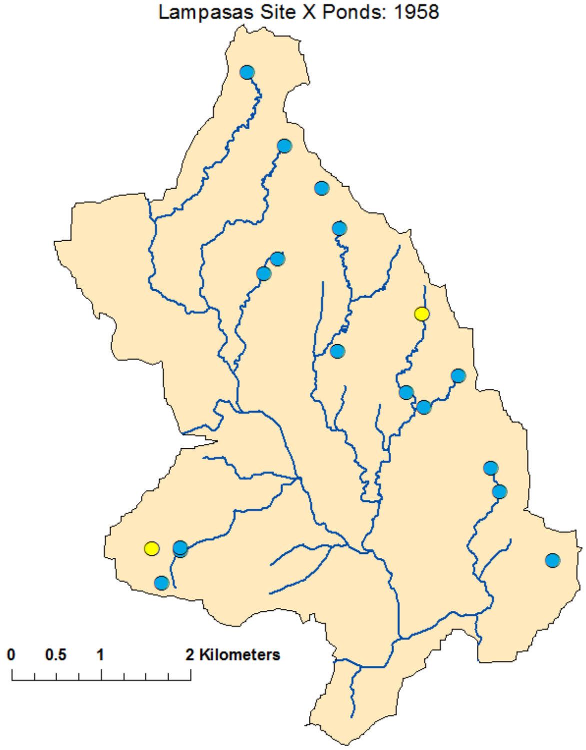
Lampasas Site 9 Ponds: Summary



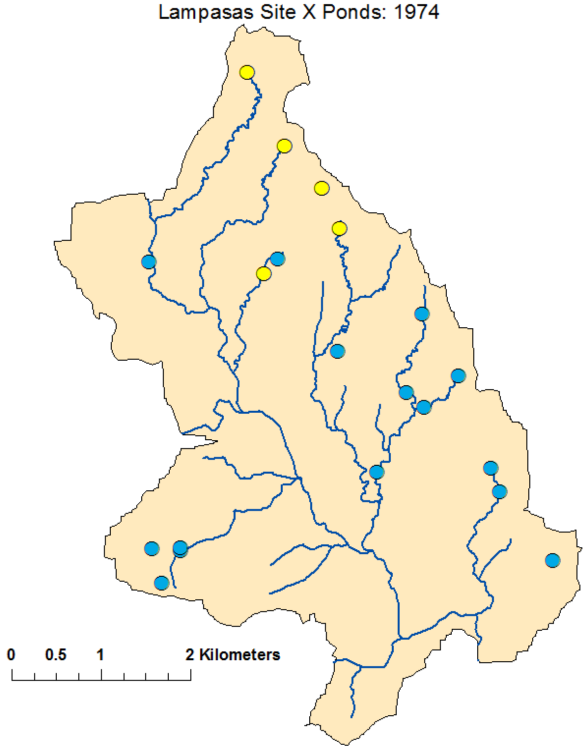
Lampasas Site X Ponds: 1941



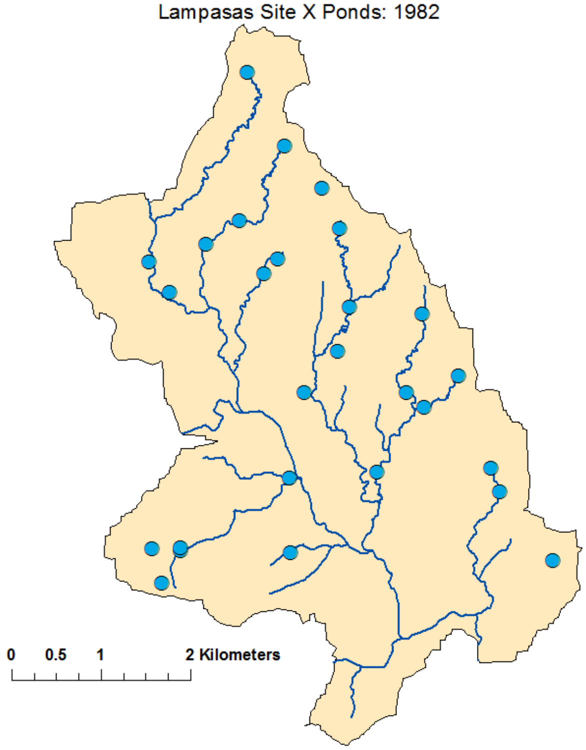
Lampasas Site X Ponds: 1958



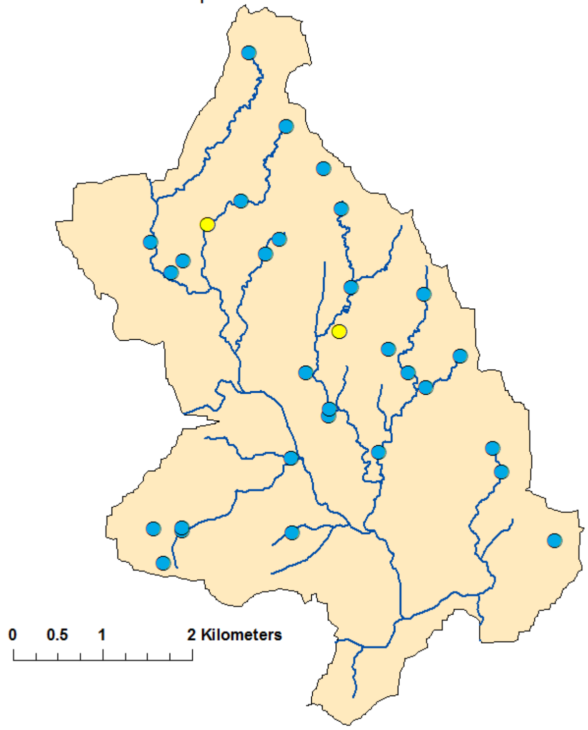
Lampasas Site X Ponds: 1974



Lampasas Site X Ponds: 1982



Lampasas Site X Ponds: 1996



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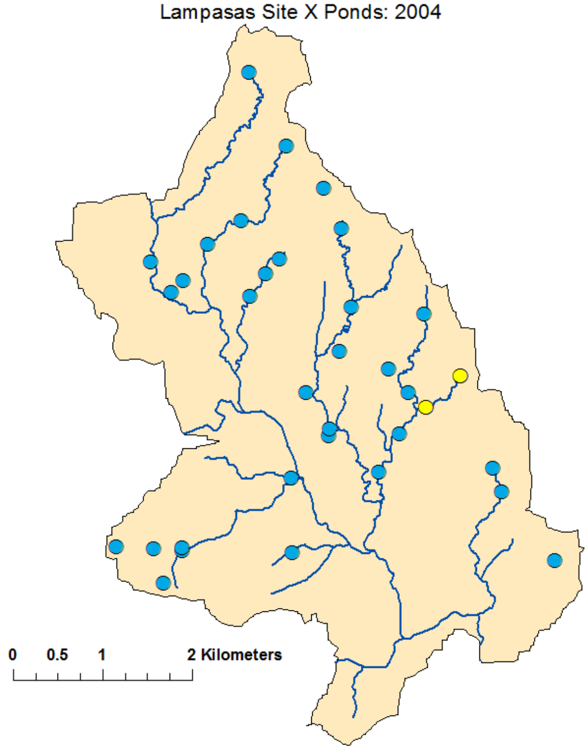
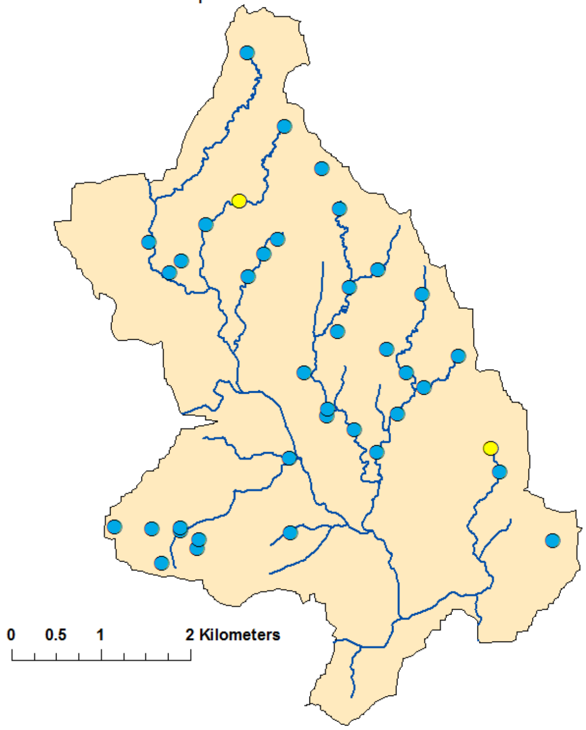


Lampasas Site X Ponds: 2008



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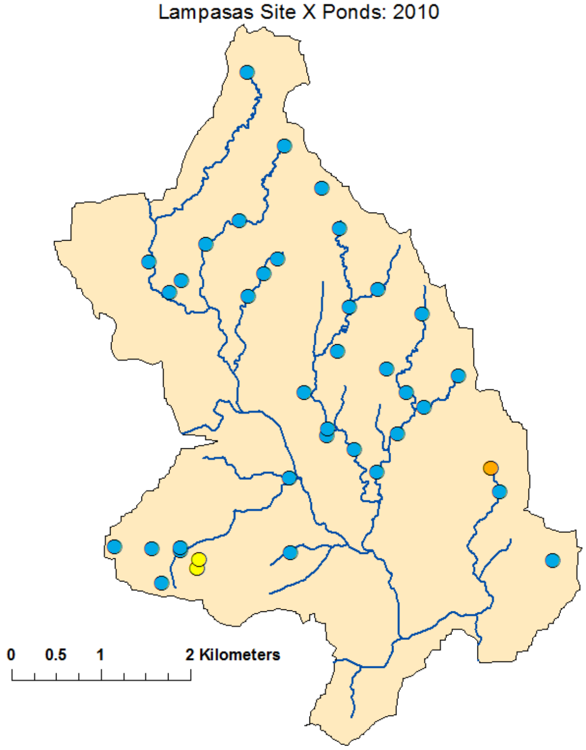

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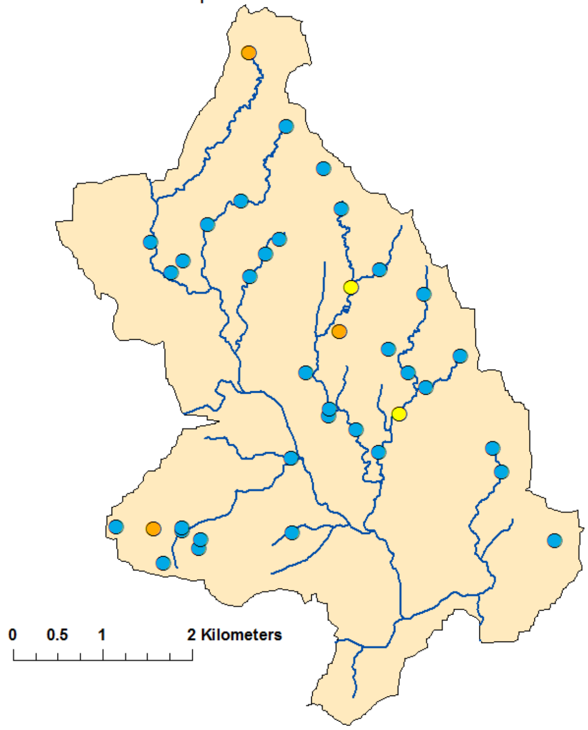
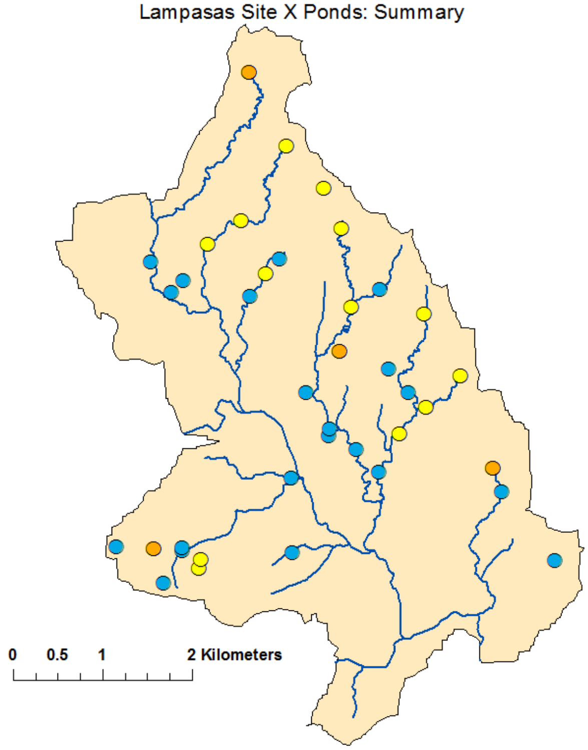
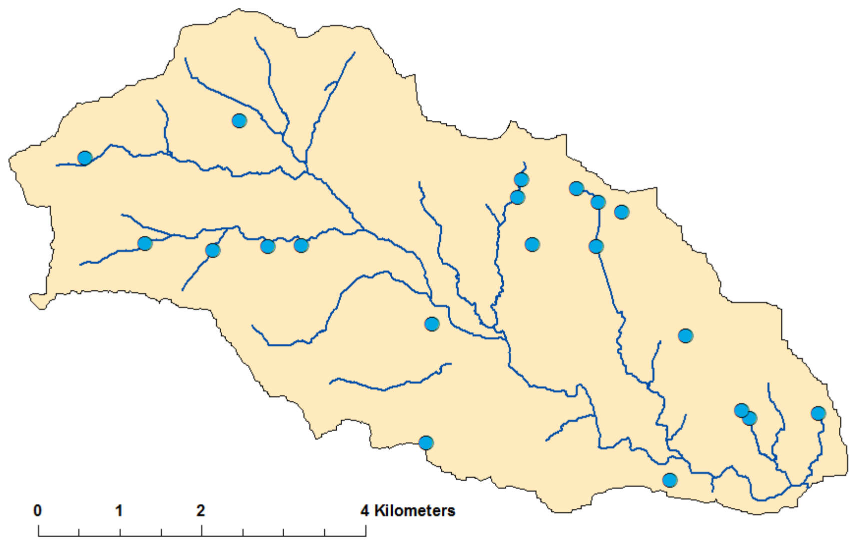


Lampasas Site X Ponds: Summary



Mills Site 1 Ponds: 1937



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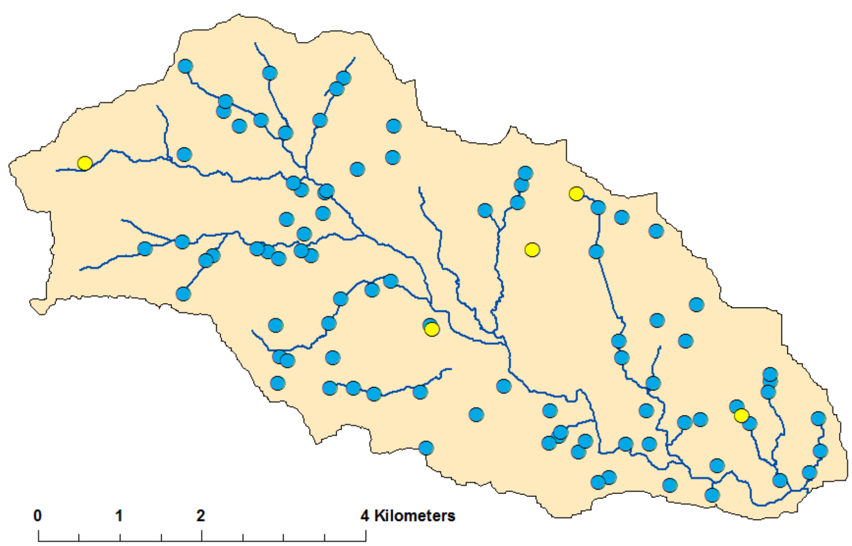

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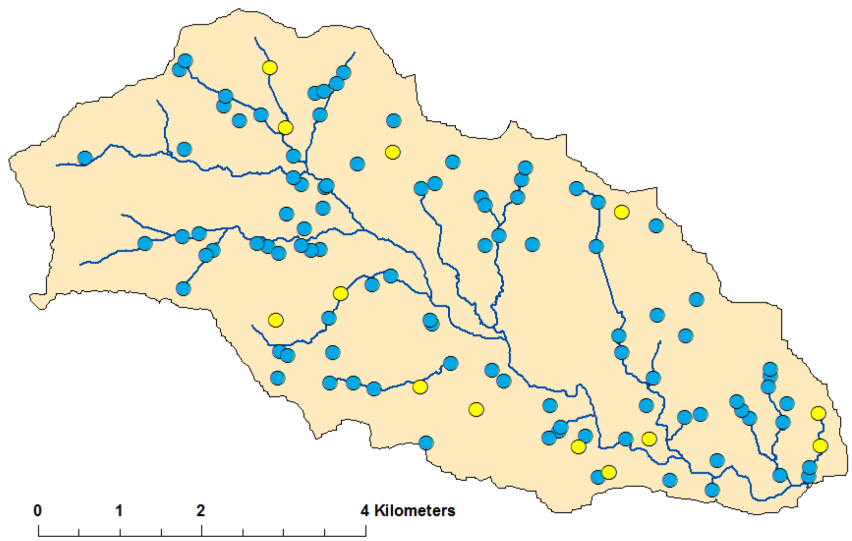
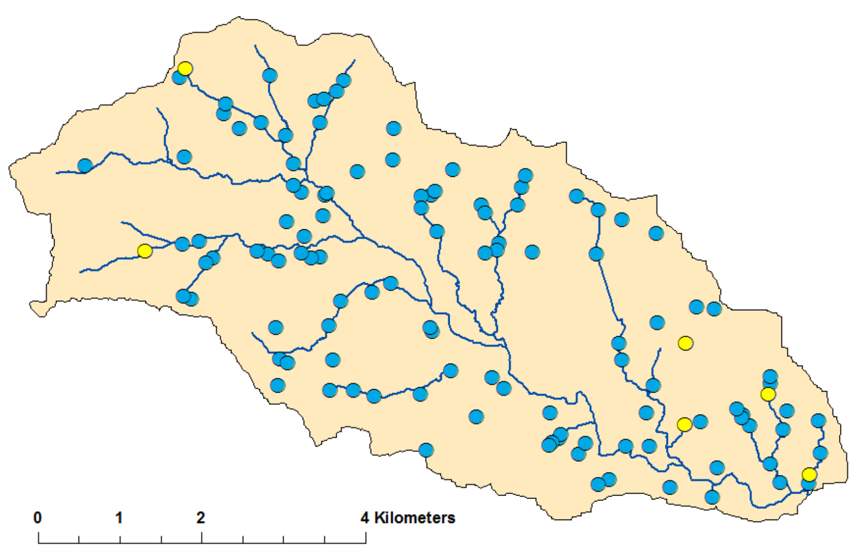


Mills Site 1 Ponds: 1982



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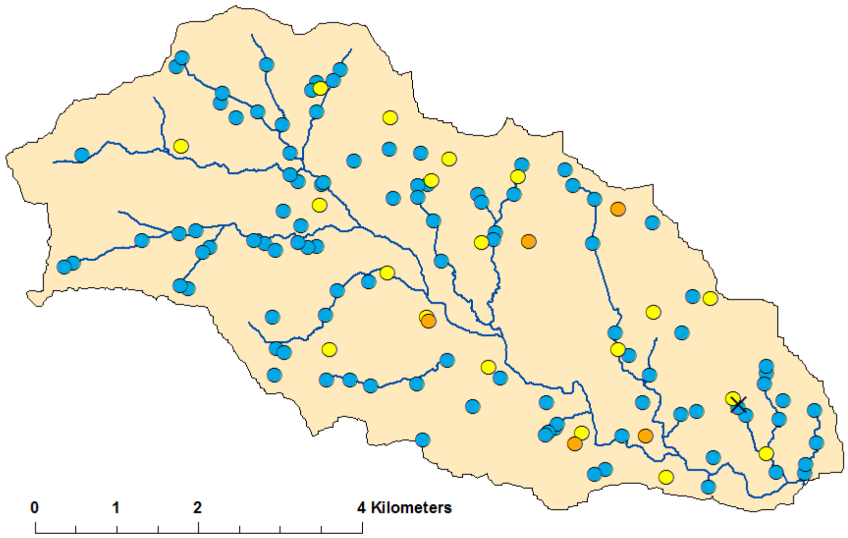

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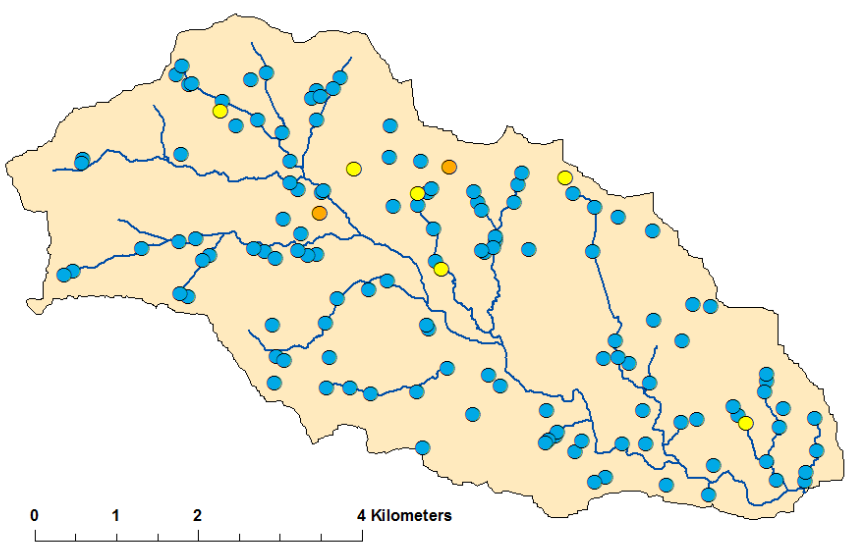

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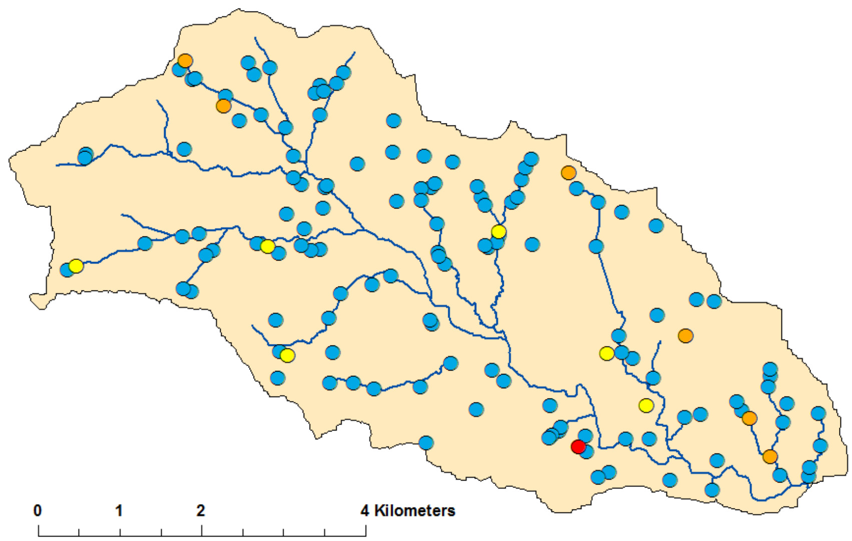

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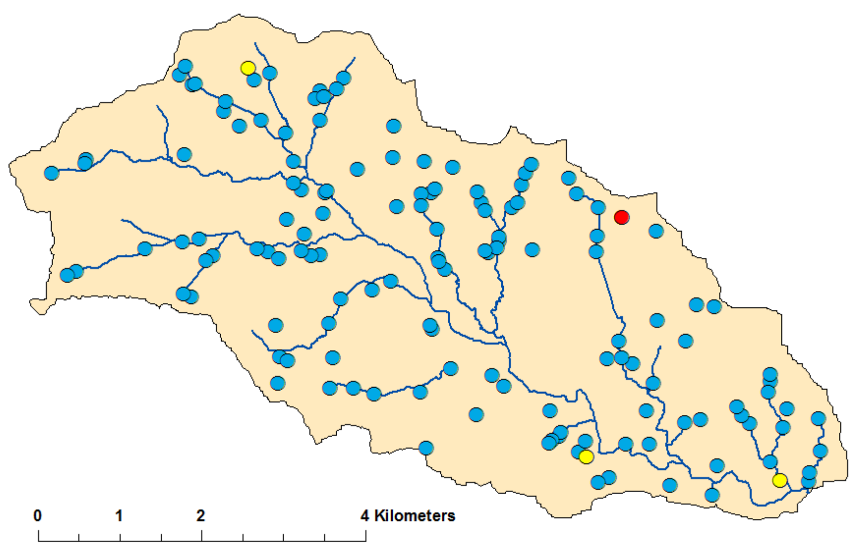

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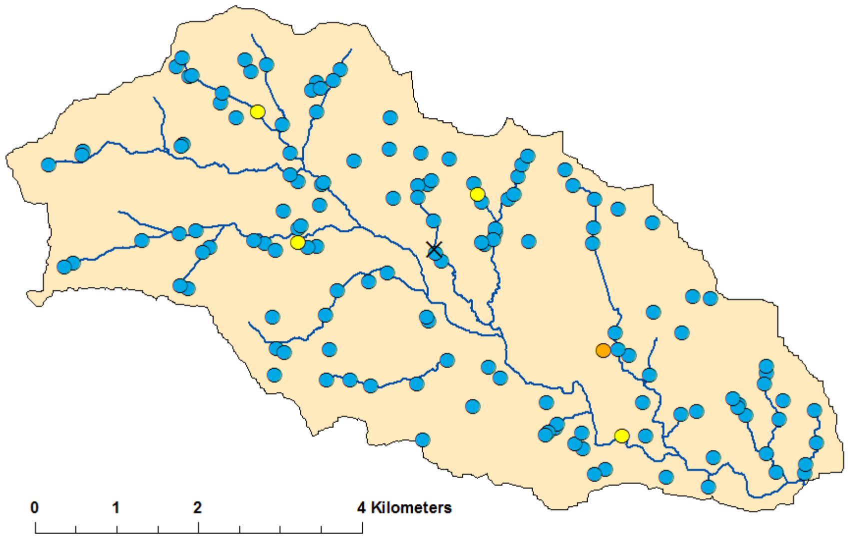

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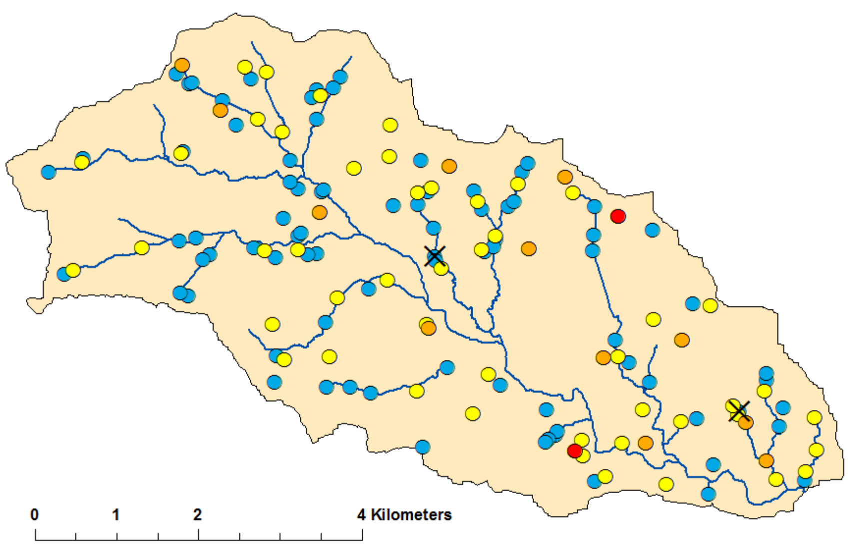
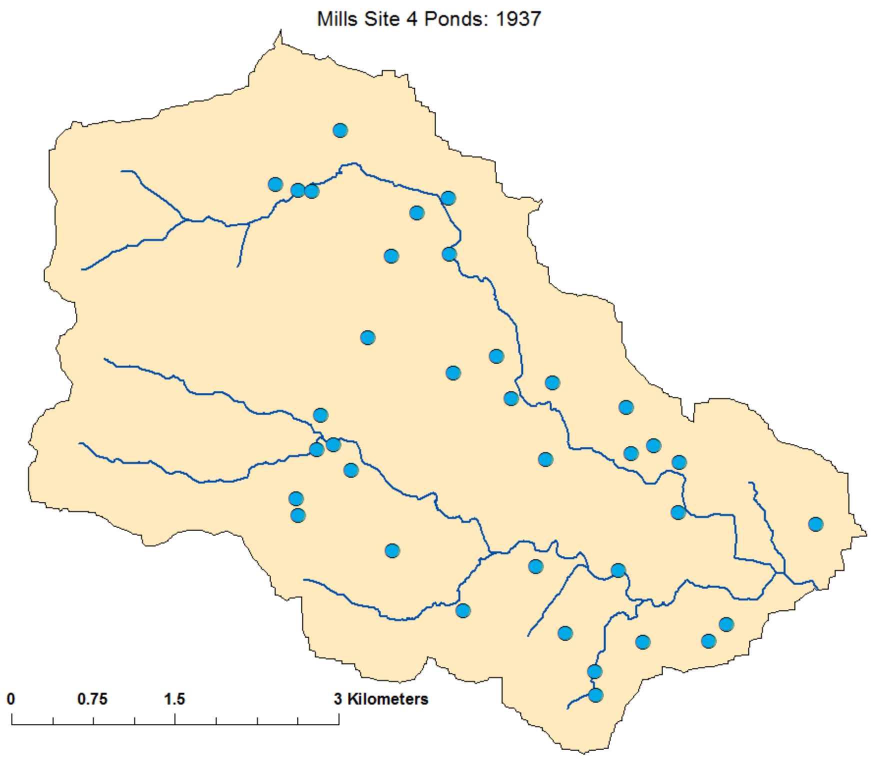
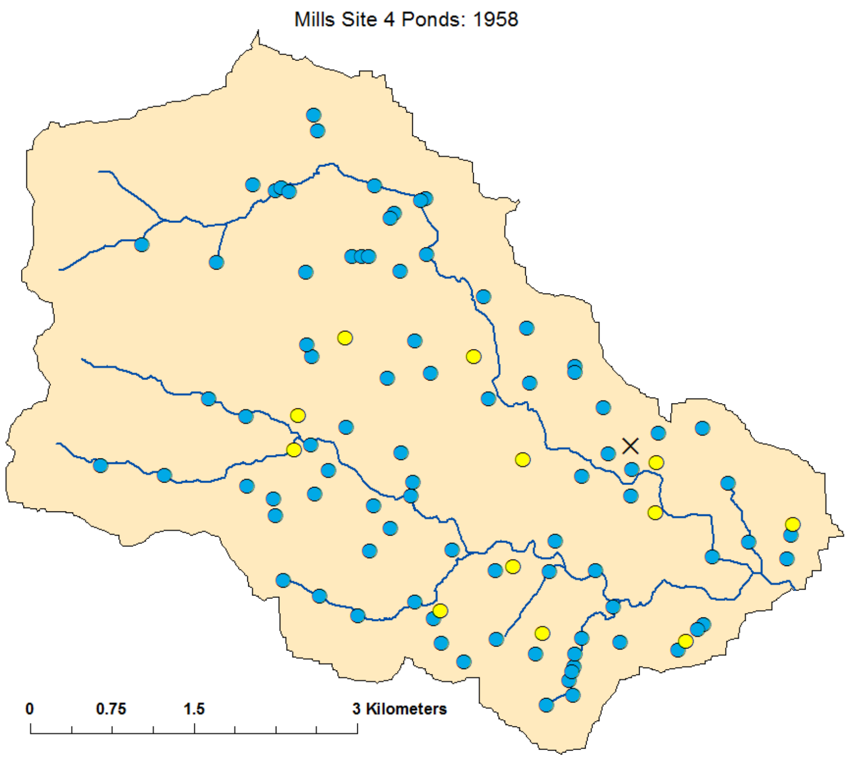
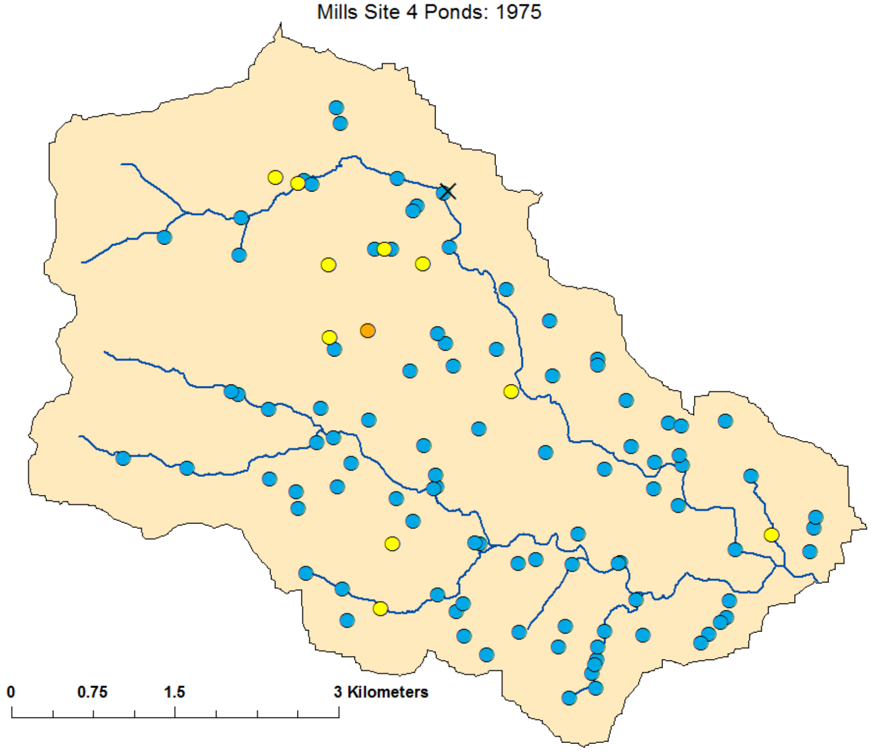



Mills Site 4 Ponds: 1937
0
0.75
1.5
3 Kilometers


Mills Site 4 Ponds: 1958
0
0.75
1.5
3 Kilometers

Mills Site 4 Ponds: 1975



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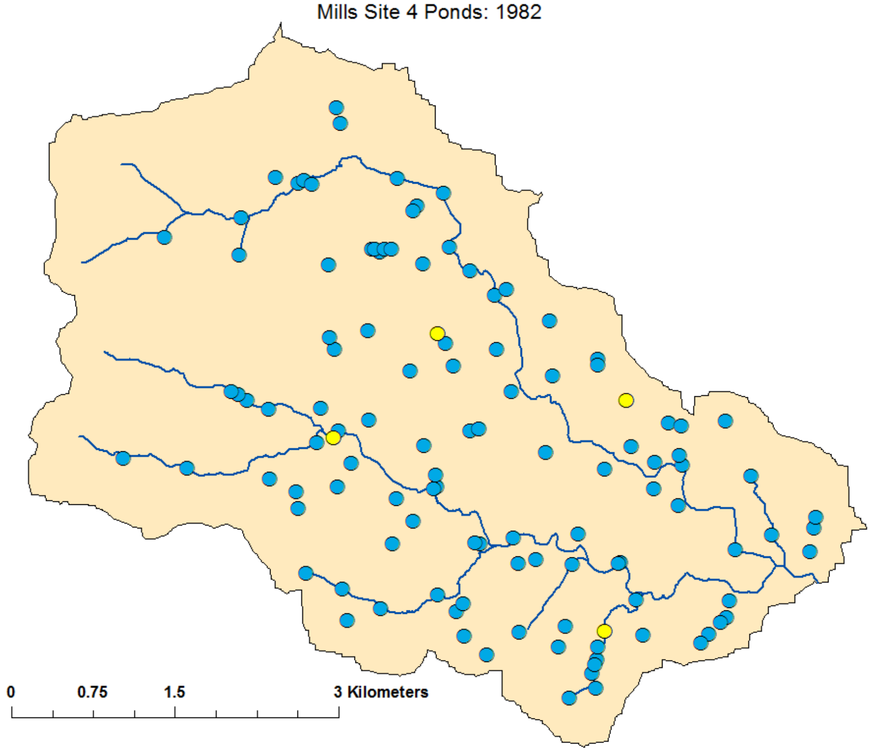

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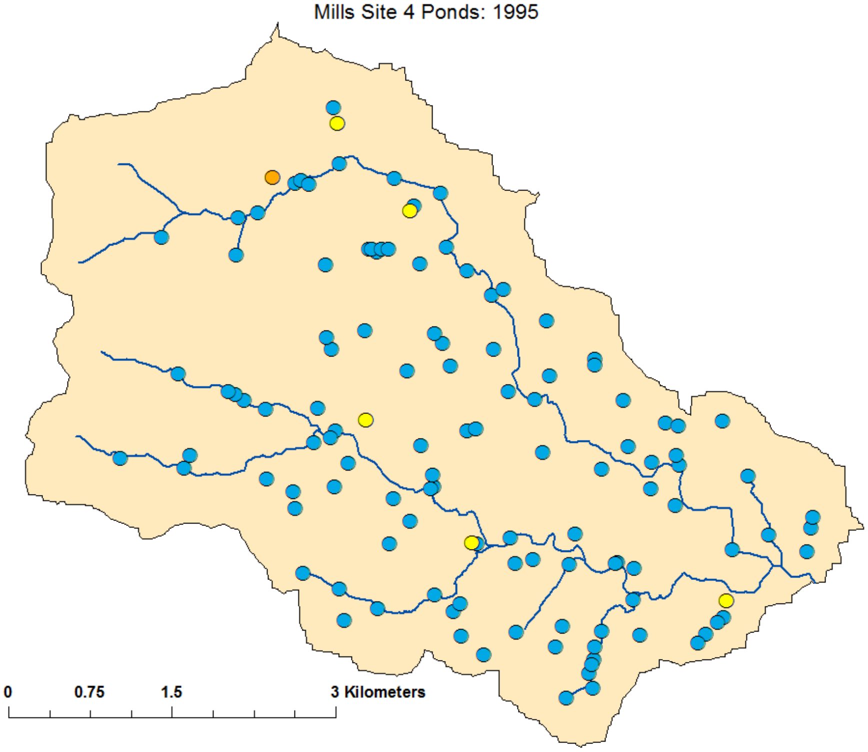

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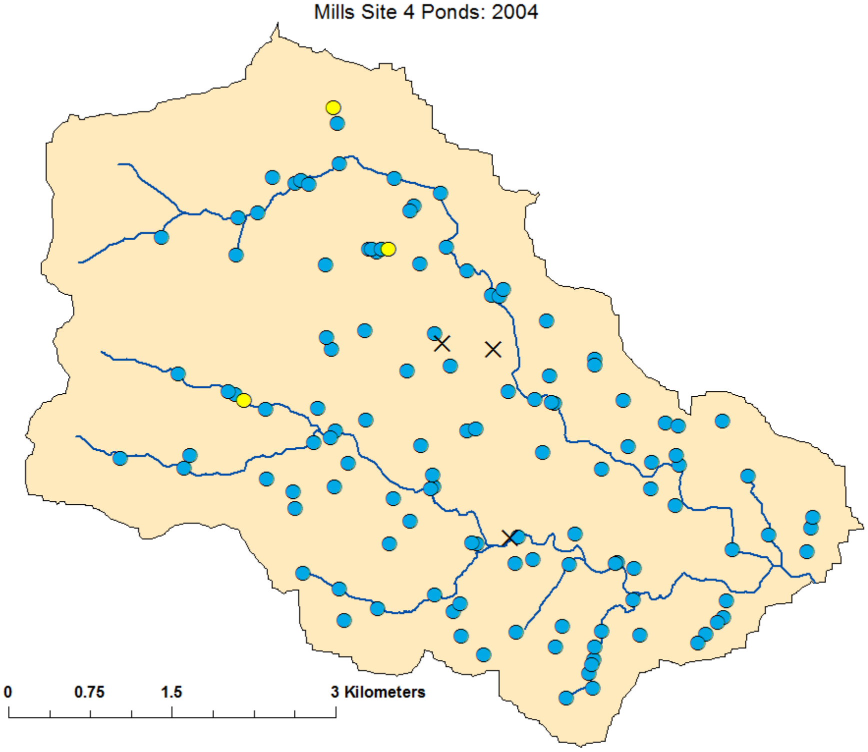

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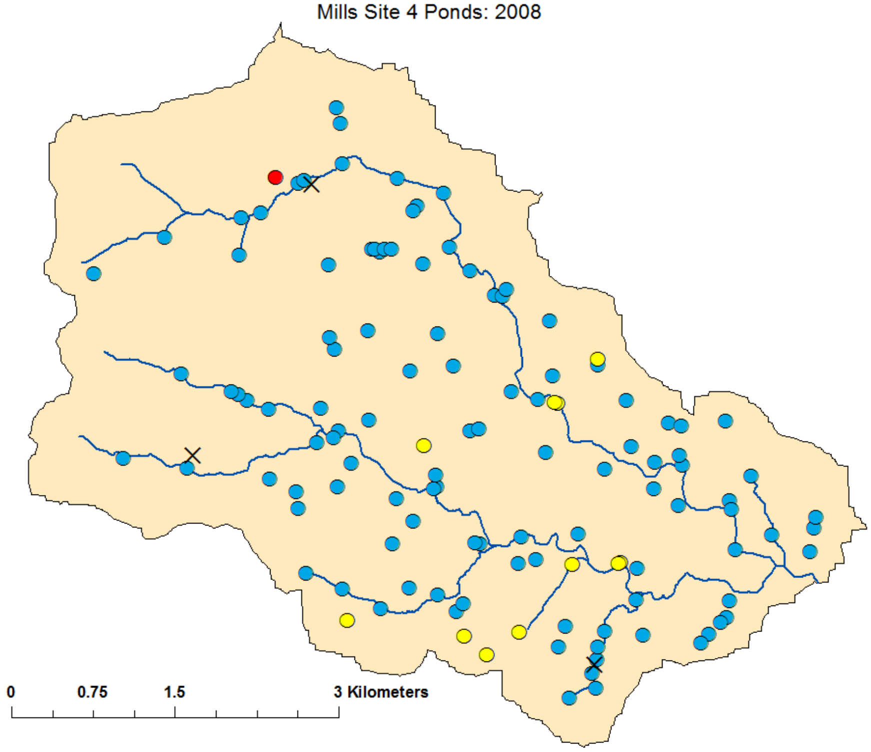

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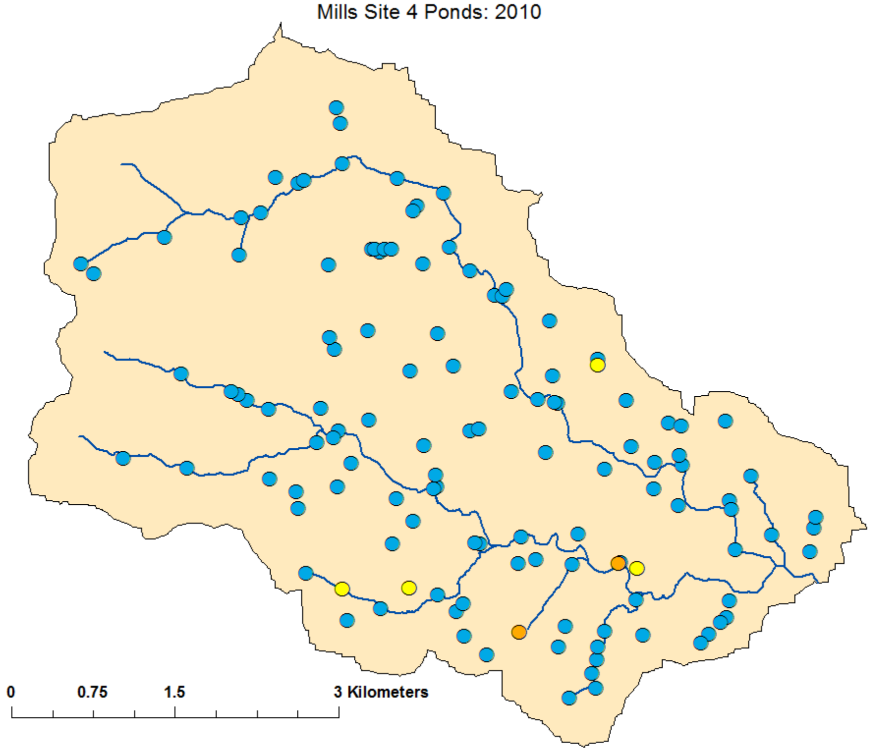
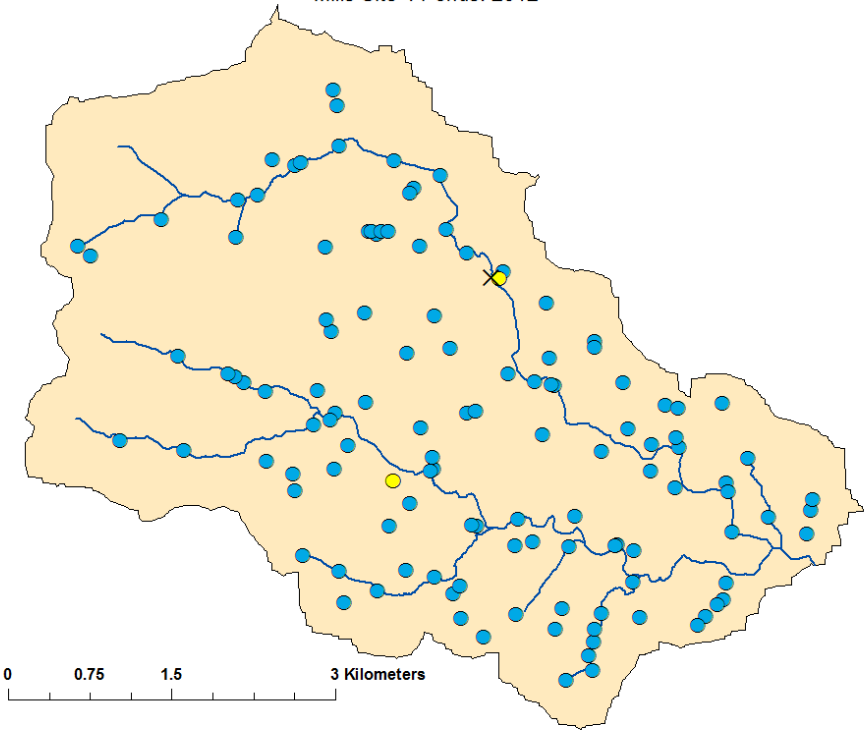
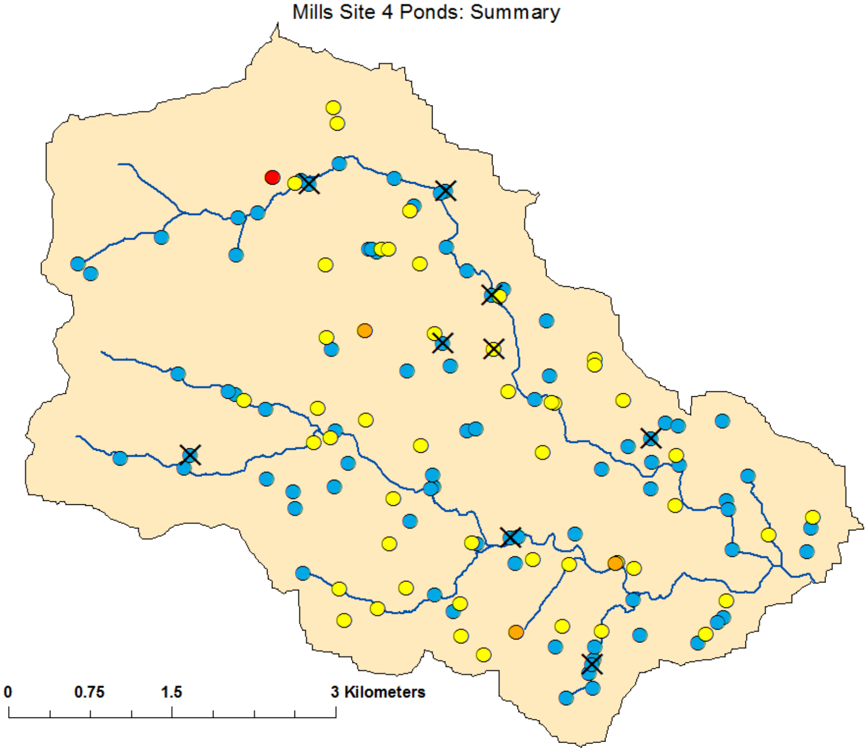


Mills Site 4 Ponds: 2012

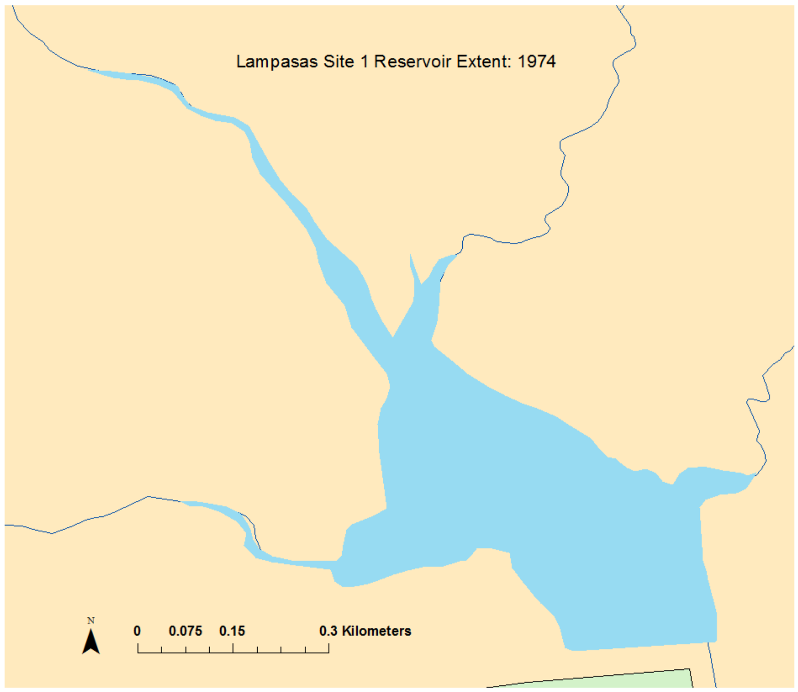


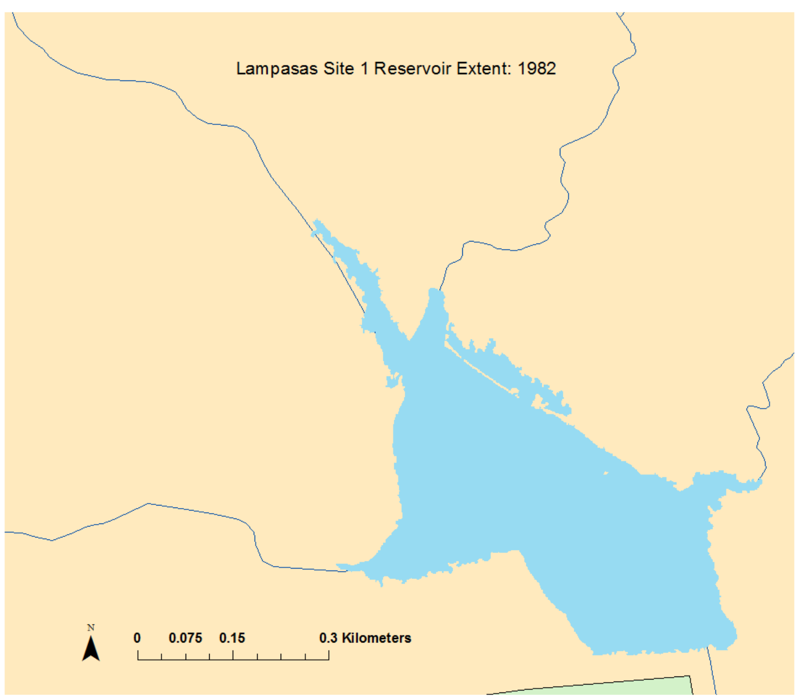
Mills Site 4 Ponds: Summary

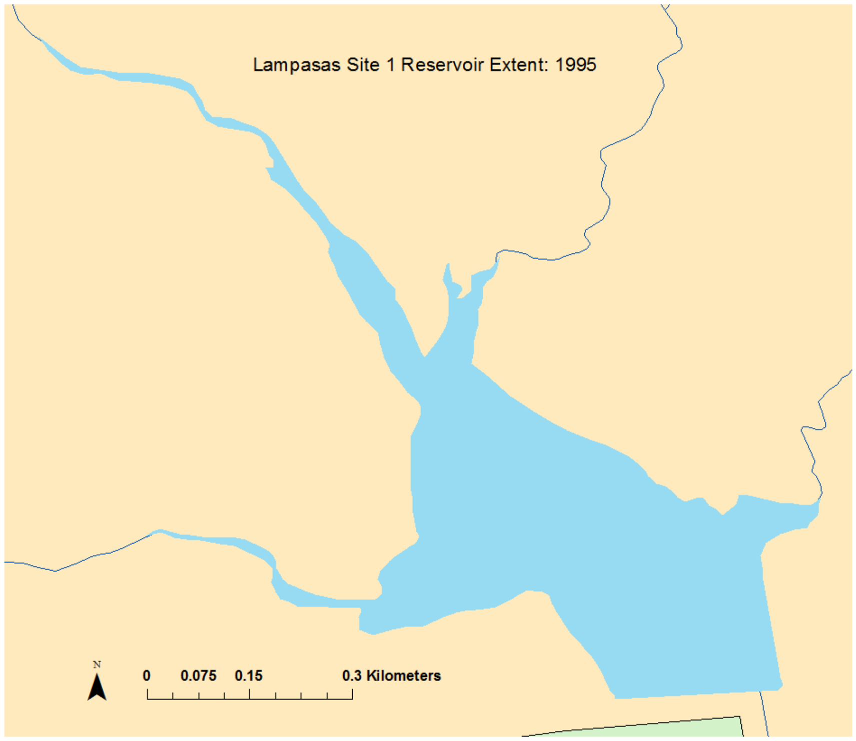


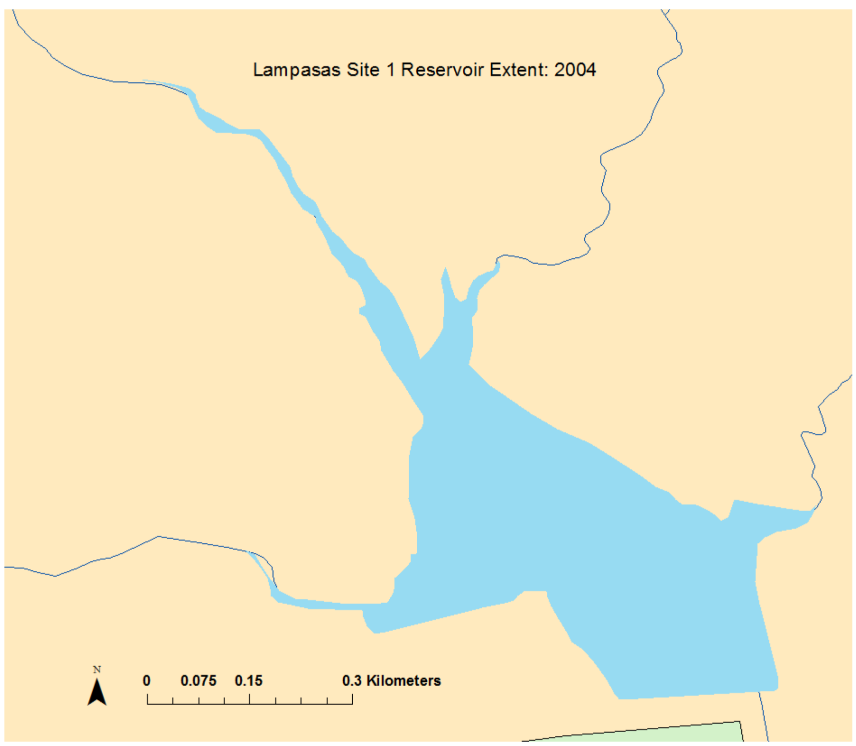
APPENDIX C

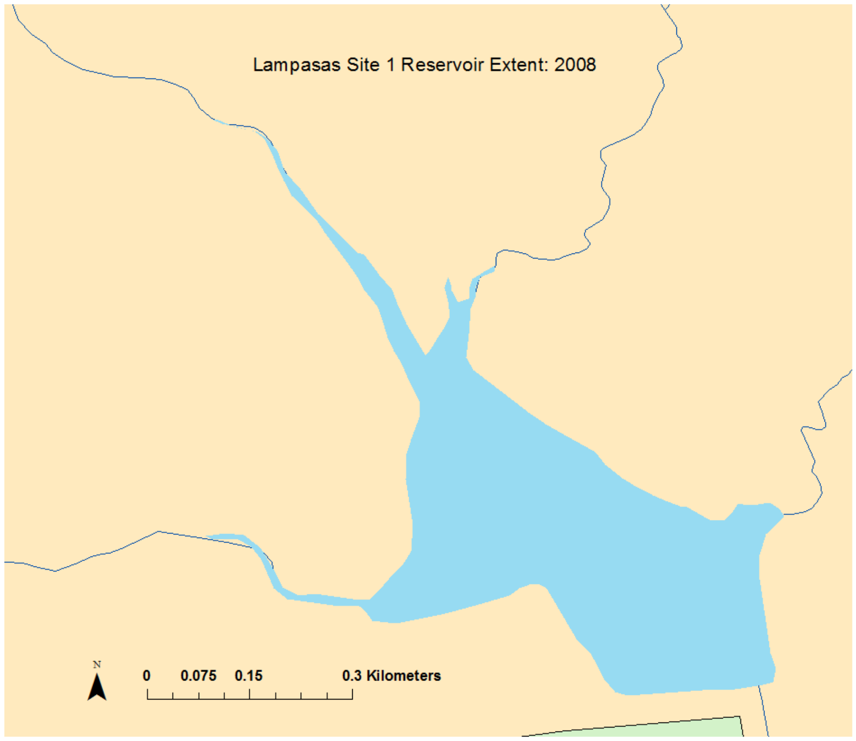
FLOOD CONTROL RESERVOIR EXTENT BY YEAR

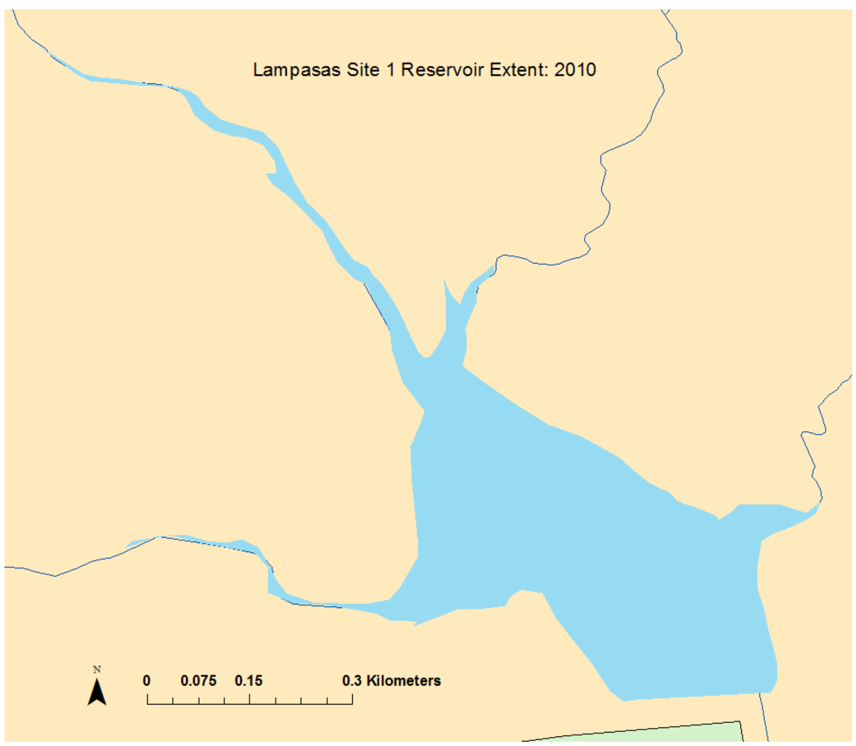


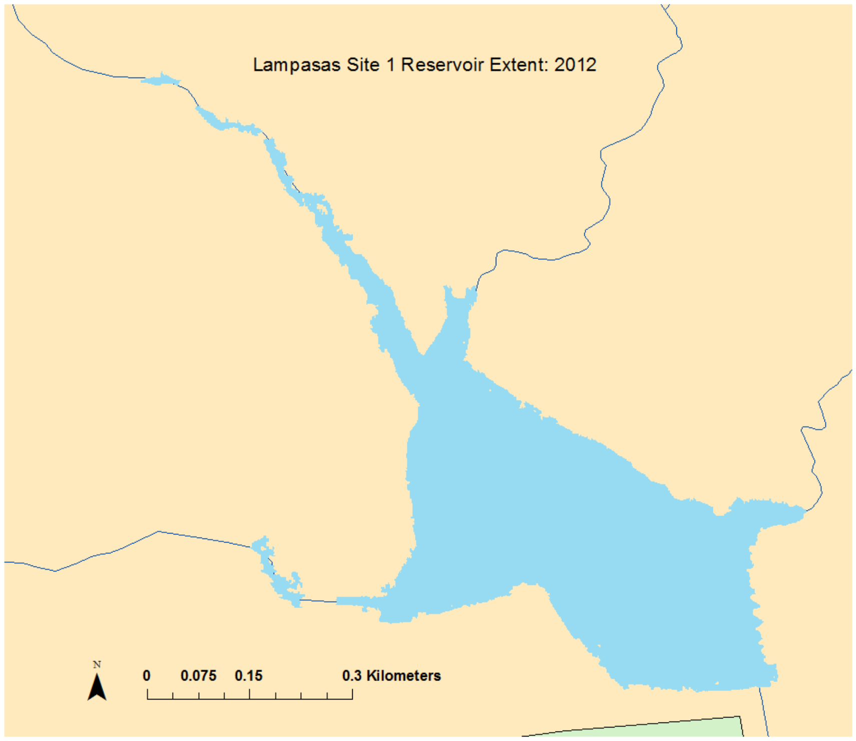


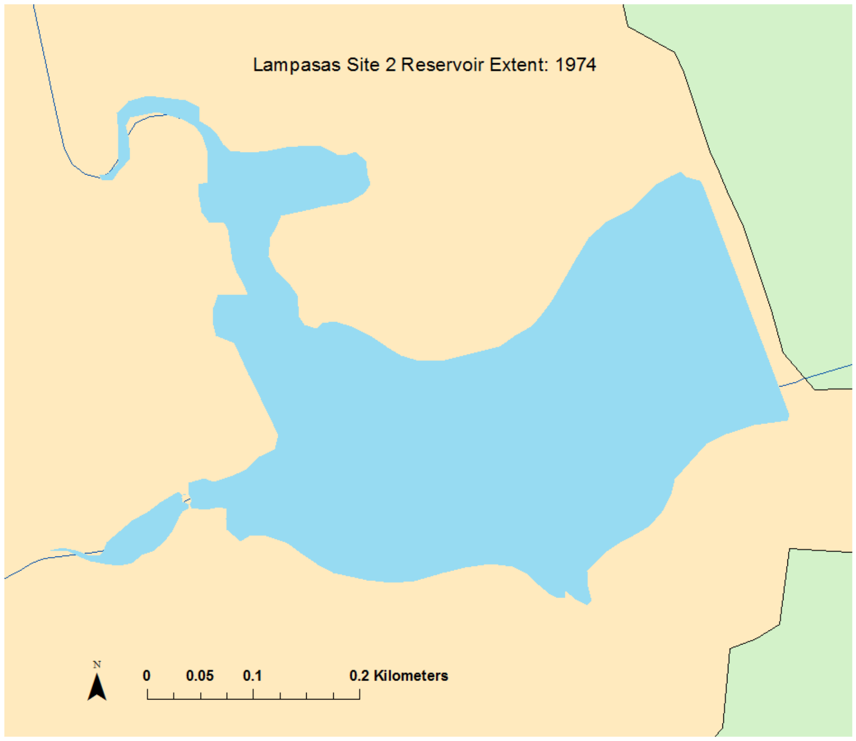

Lampasas Site 1 Reservoir Extent: 1995
N
0 0.075 0.15 0.3 Kilometers

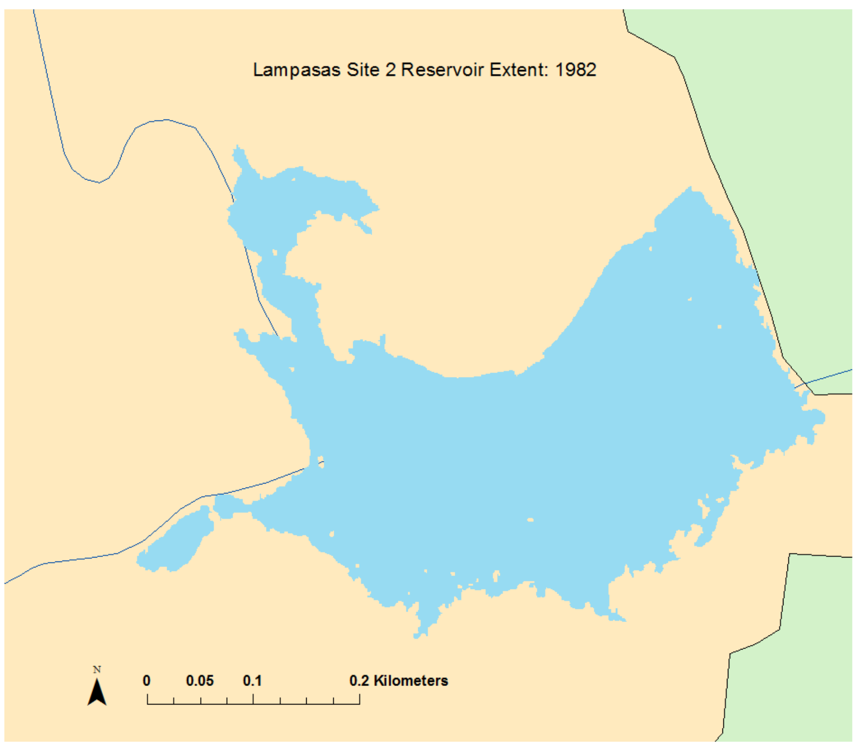

Lampasas Site 1 Reservoir Extent: 2004
N
0 0.075 0.15 0.3 Kilometers

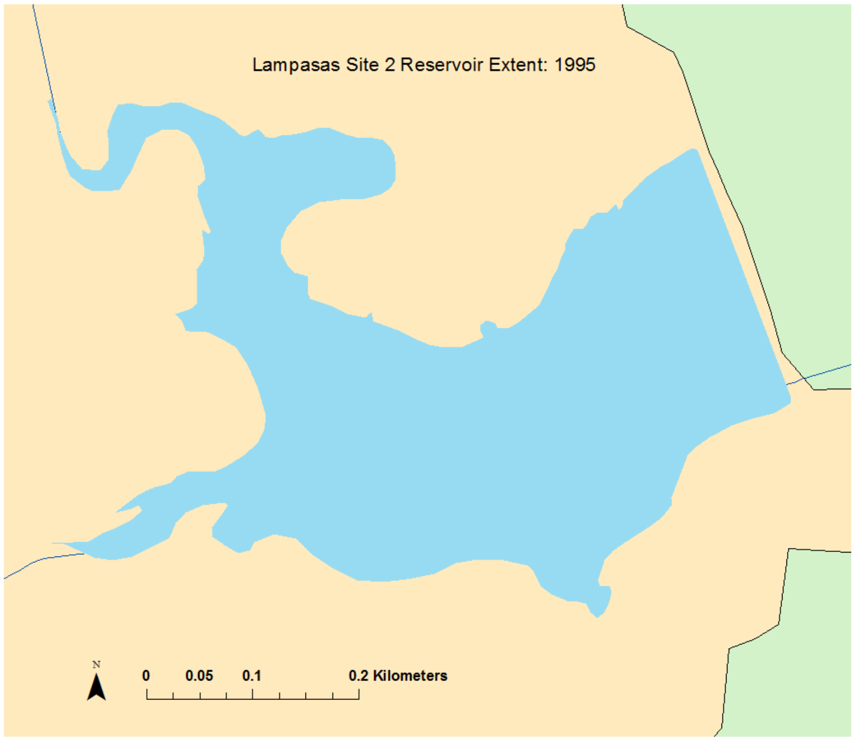

Lampasas Site 1 Reservoir Extent: 2008
N
0 0.075 0.15 0.3 Kilometers

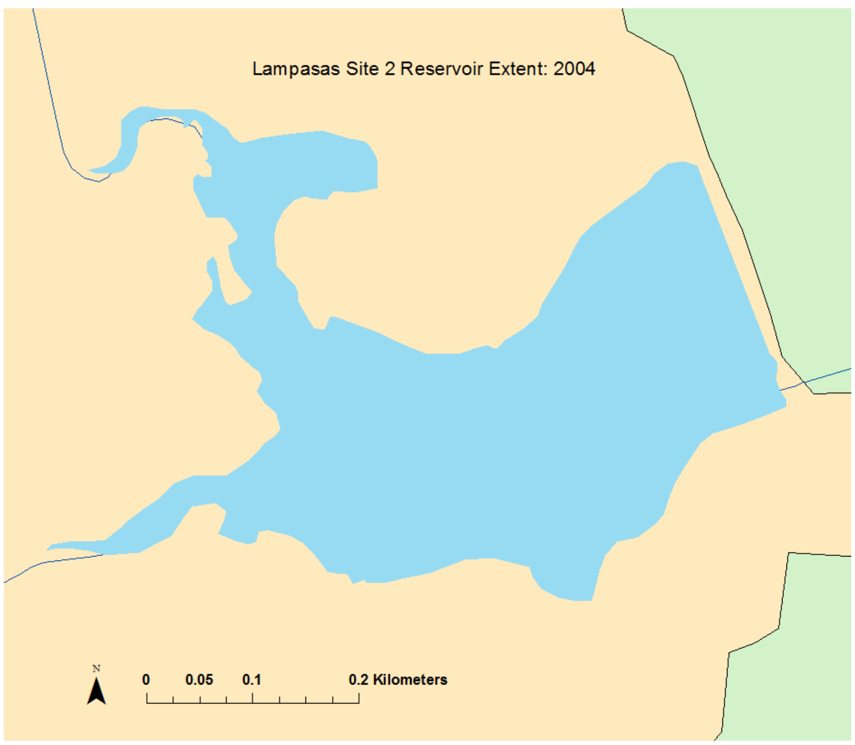

Lampasas Site 1 Reservoir Extent: 2010
N
0 0.075 0.15 0.3 Kilometers

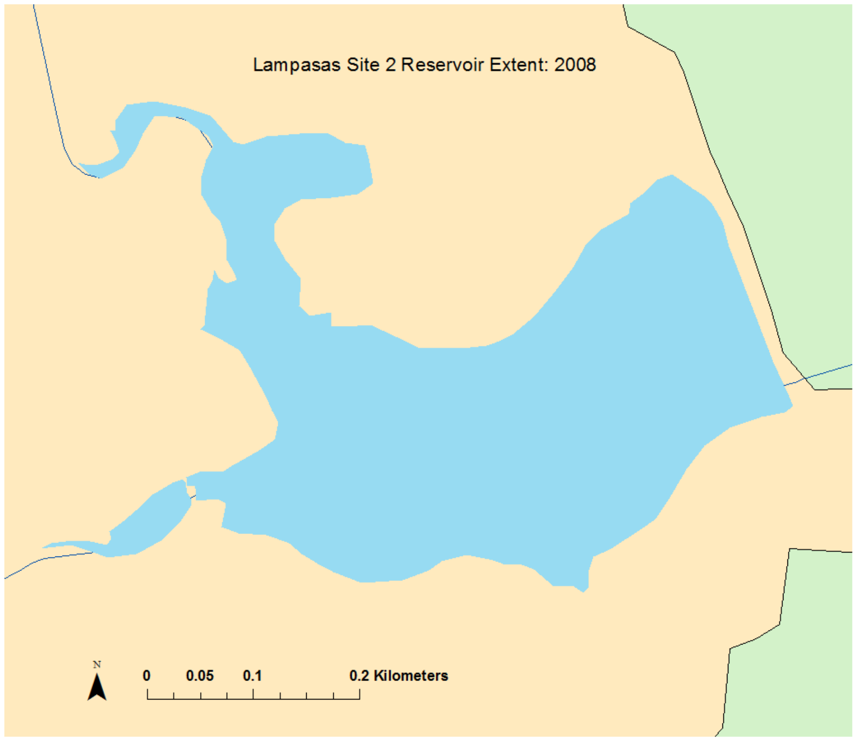

Lampasas Site 1 Reservoir Extent: 2012
N
0
0.075
0.15
0.3 Kilometers

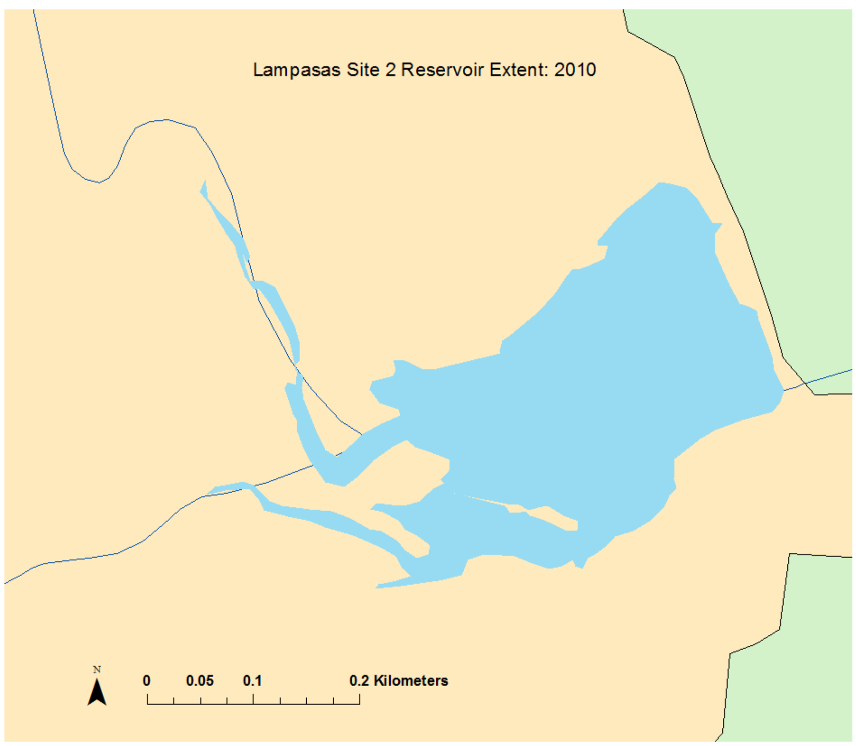

Lampasas Site 2 Reservoir Extent: 1974
N
0 0.05 0.1 0.2 Kilometers

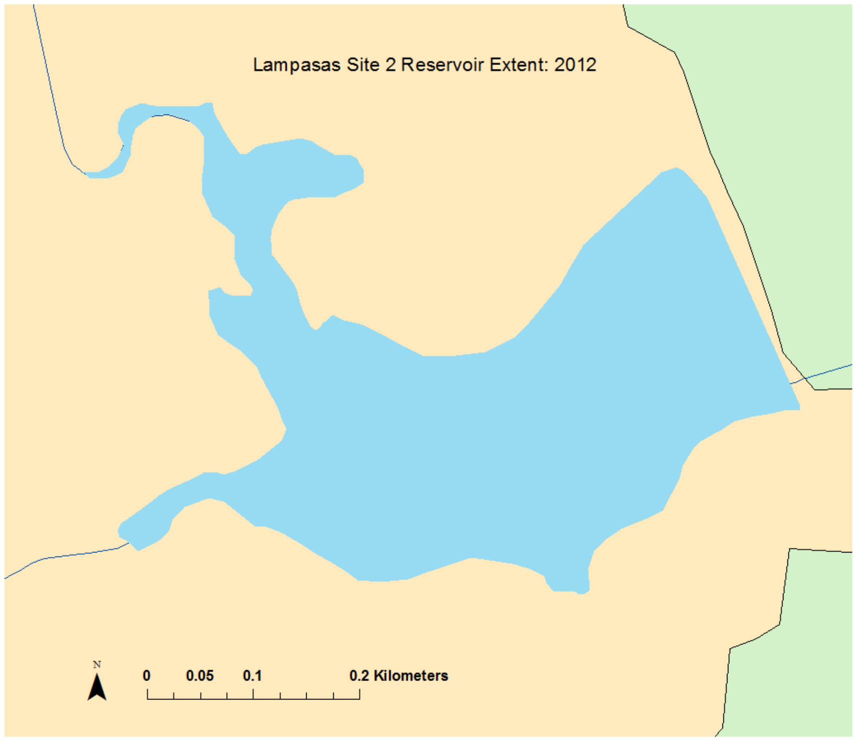

Lampasas Site 2 Reservoir Extent: 1982
N
0 0.05 0.1 0.2 Kilometers

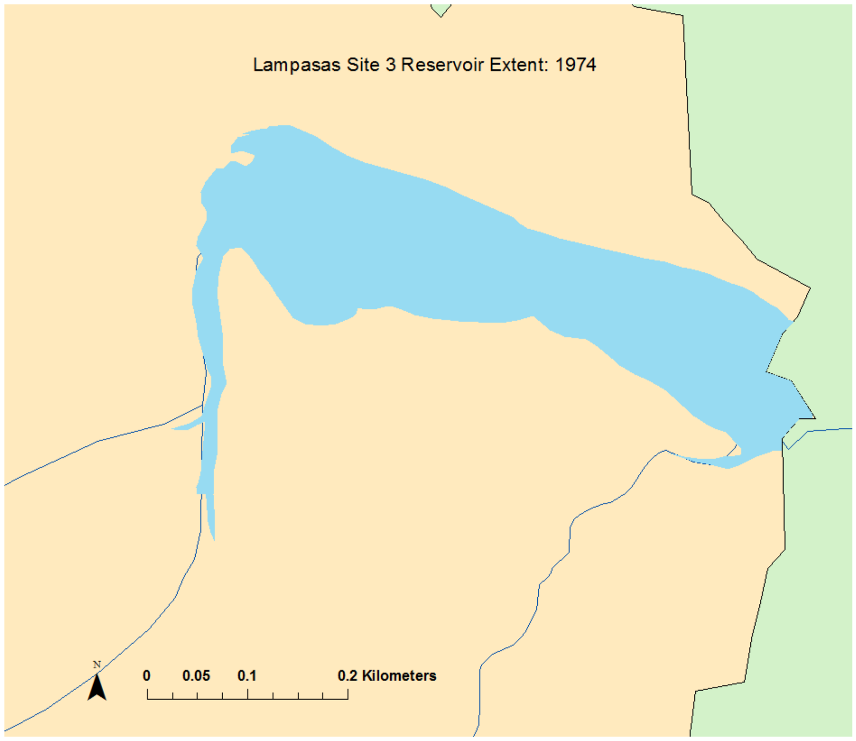

Lampasas Site 2 Reservoir Extent: 1995
N
0 0.05 0.1 0.2 Kilometers

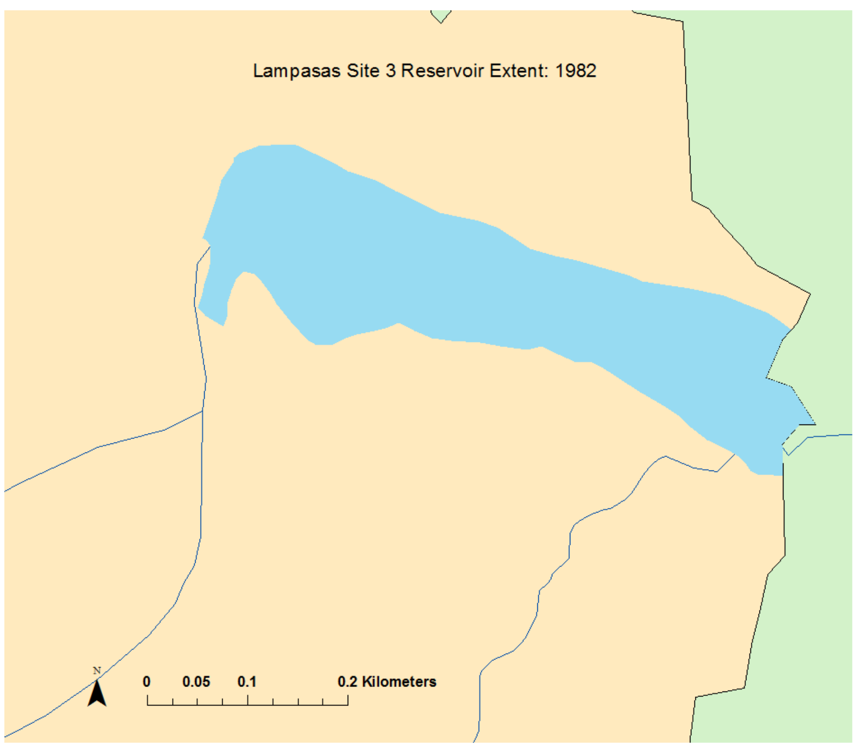

Lampasas Site 2 Reservoir Extent: 2004
N
0 0.05 0.1 0.2 Kilometers

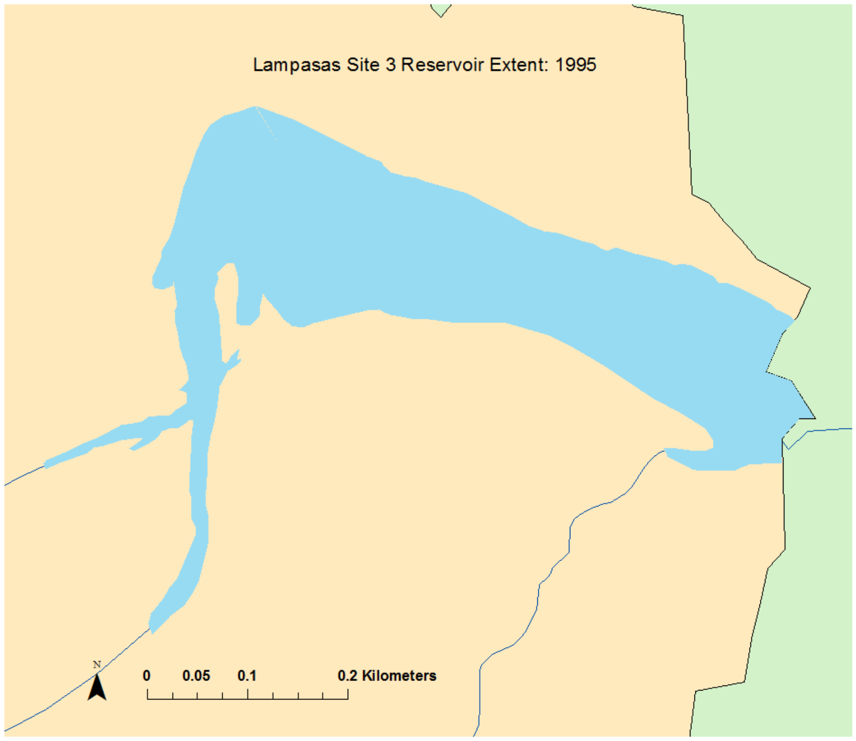

Lampasas Site 2 Reservoir Extent: 2008
N
0
0.05
0.1
0.2 Kilometers

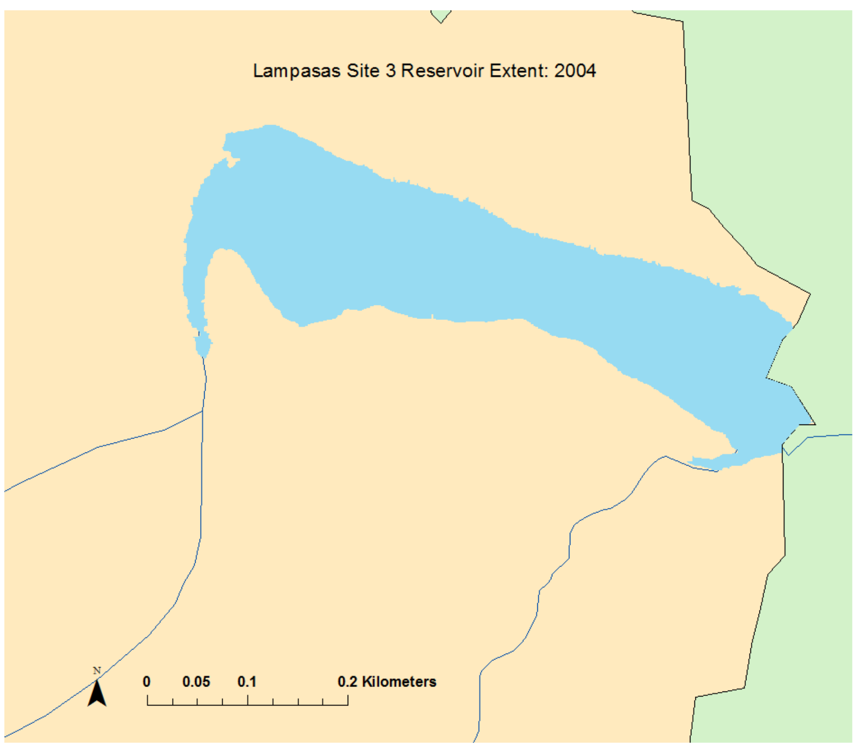

Lampasas Site 2 Reservoir Extent: 2010
N
0
0.05
0.1
0.2 Kilometers

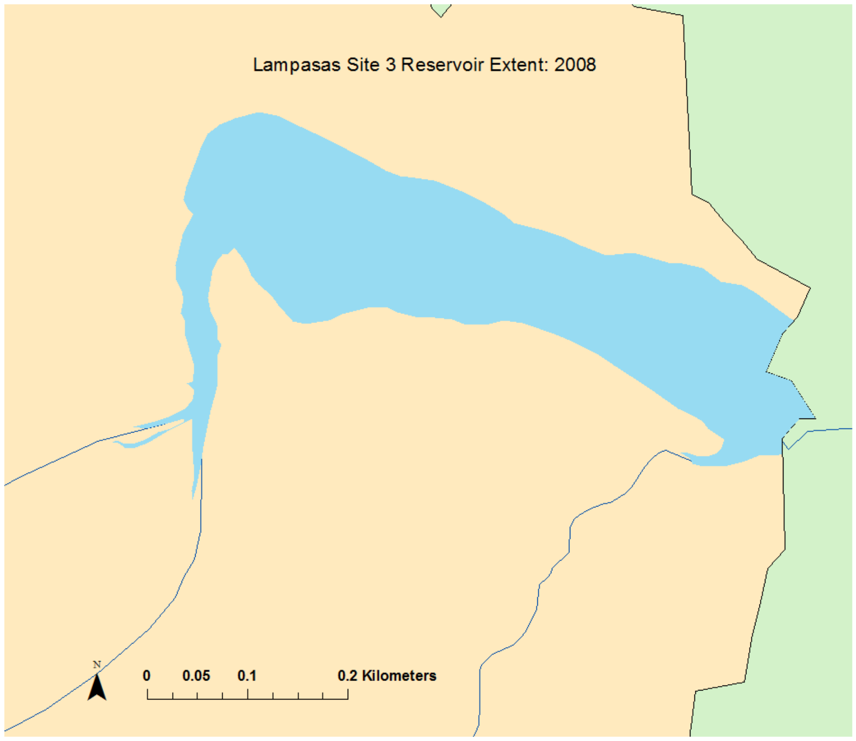

Lampasas Site 2 Reservoir Extent: 2012
N
0
0.05
0.1
0.2 Kilometers

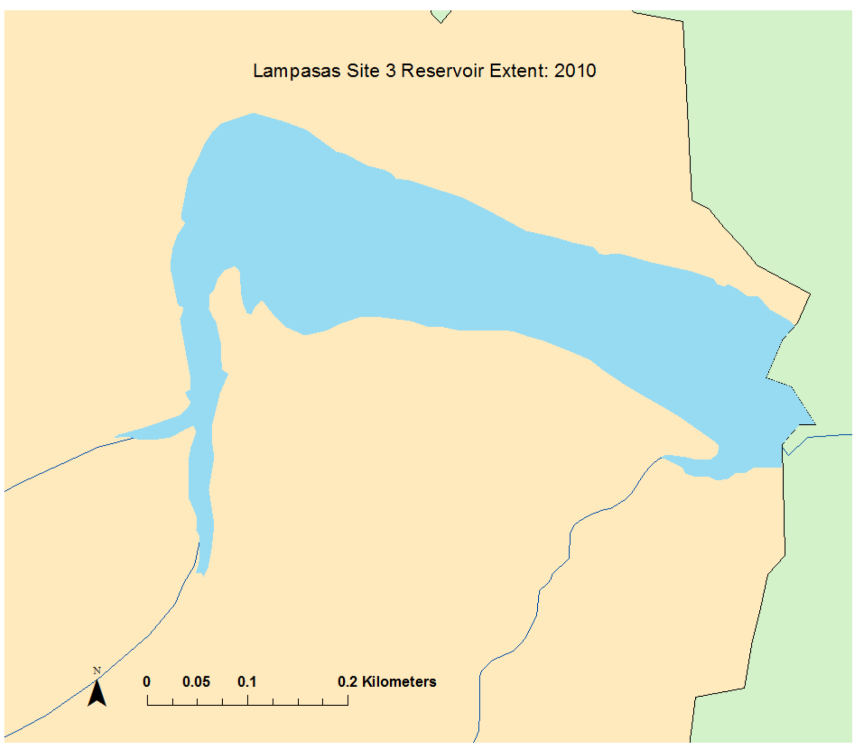

Lampasas Site 3 Reservoir Extent: 1974
N
0
0.05
0.1
0.2 Kilometers

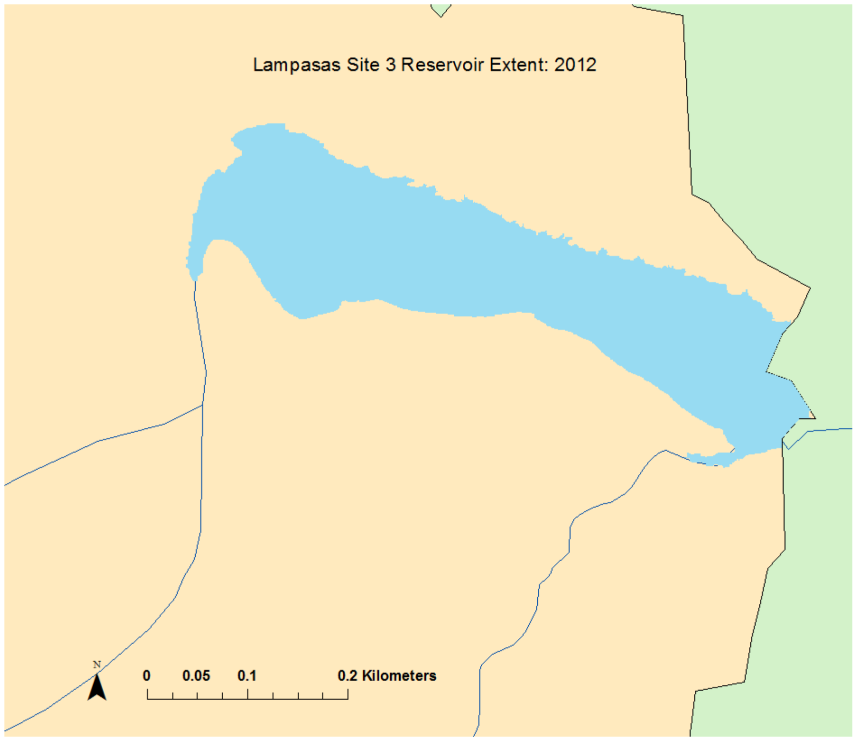

Lampasas Site 3 Reservoir Extent: 1982
N
0
0.05
0.1
0.2 Kilometers

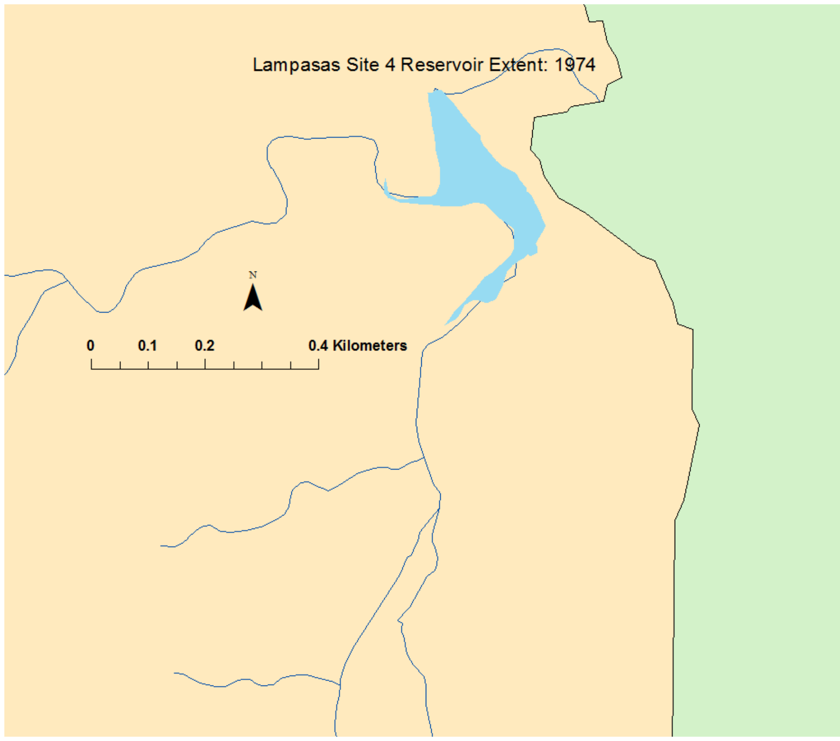

Lampasas Site 3 Reservoir Extent: 1995
N
0 0.05 0.1 0.2 Kilometers

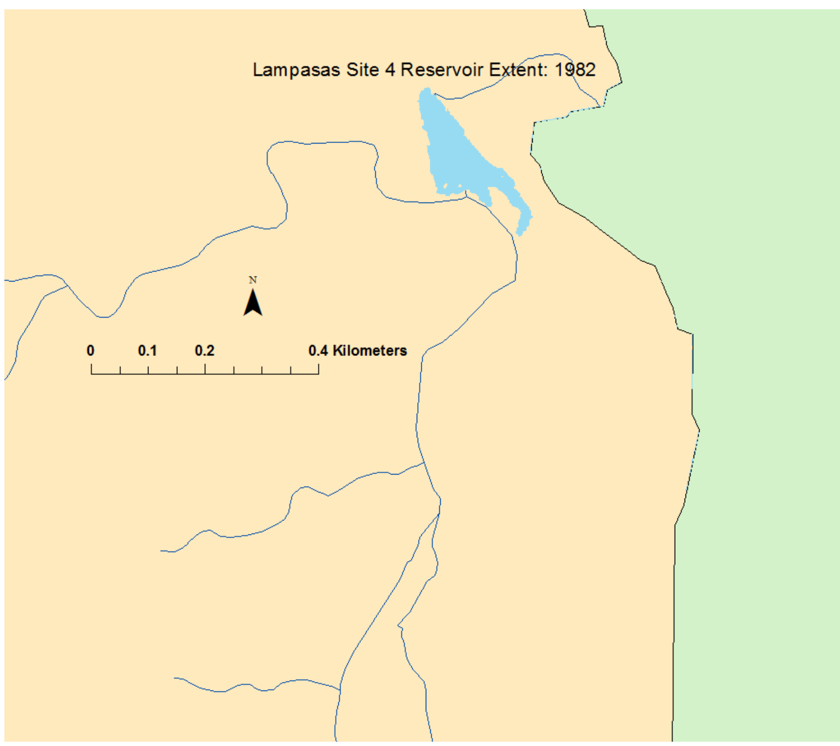

Lampasas Site 3 Reservoir Extent: 2004
N
0 0.05 0.1 0.2 Kilometers

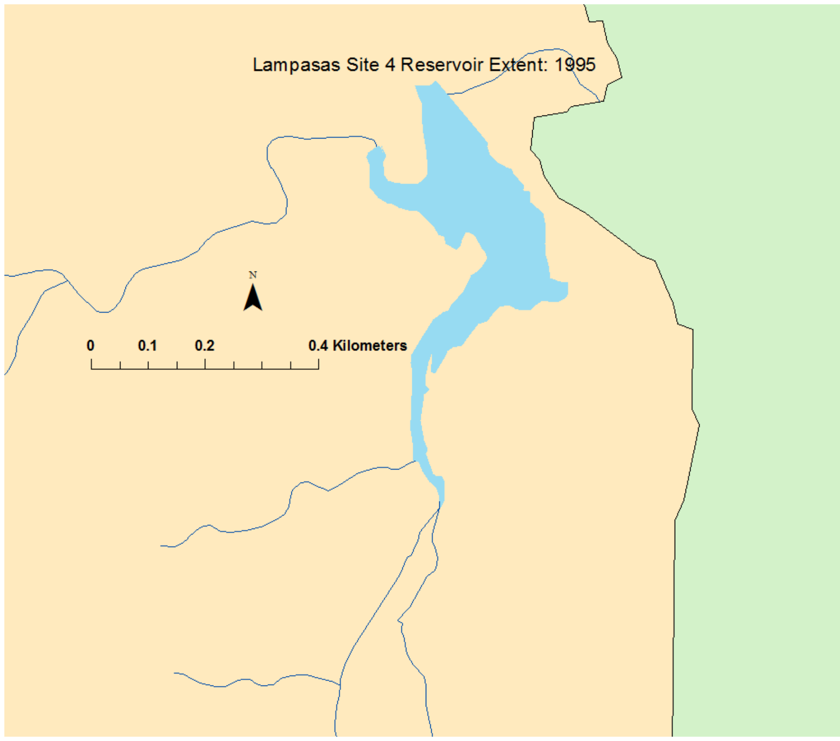

Lampasas Site 3 Reservoir Extent: 2008
N
0 0.05 0.1 0.2 Kilometers

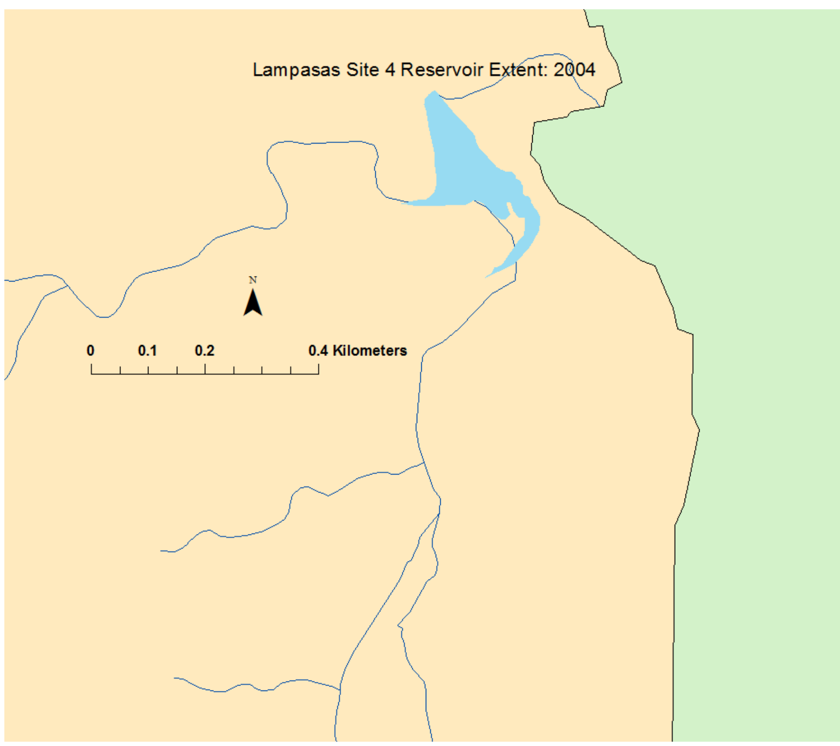

Lampasas Site 3 Reservoir Extent: 2010
N
0 0.05 0.1 0.2 Kilometers

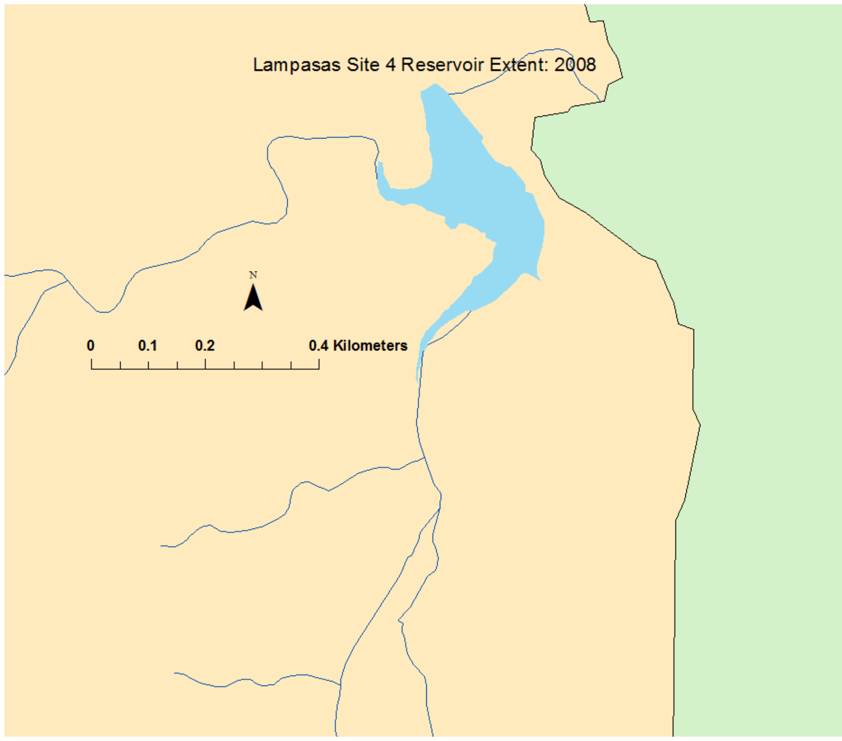

Lampasas Site 3 Reservoir Extent: 2012
N
0 0.05 0.1 0.2 Kilometers

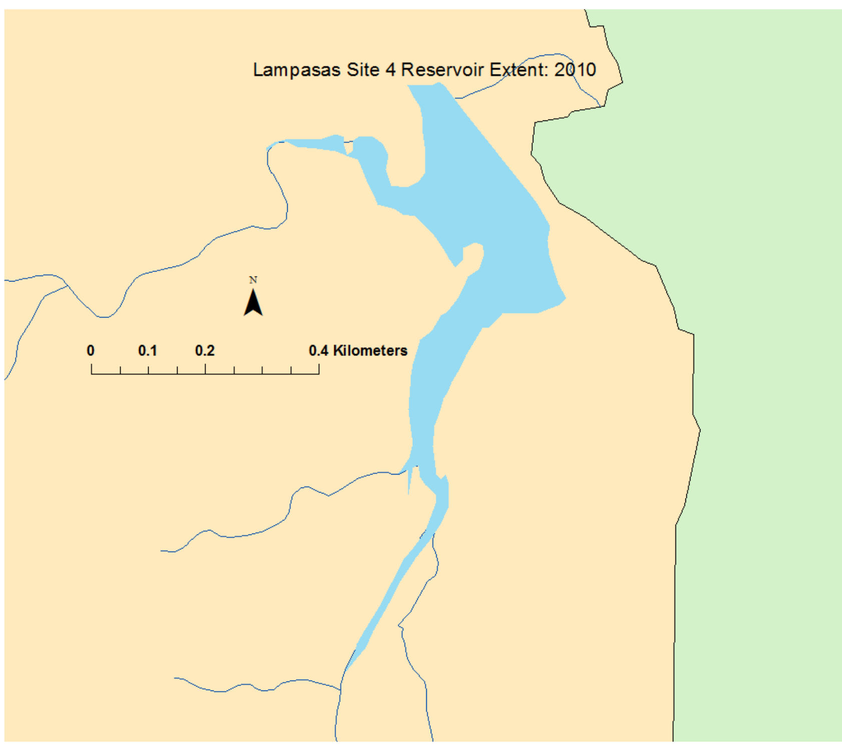

Lampasas Site 4 Reservoir Extent: 1974
N
0
0.1
0.2
0.4 Kilometers

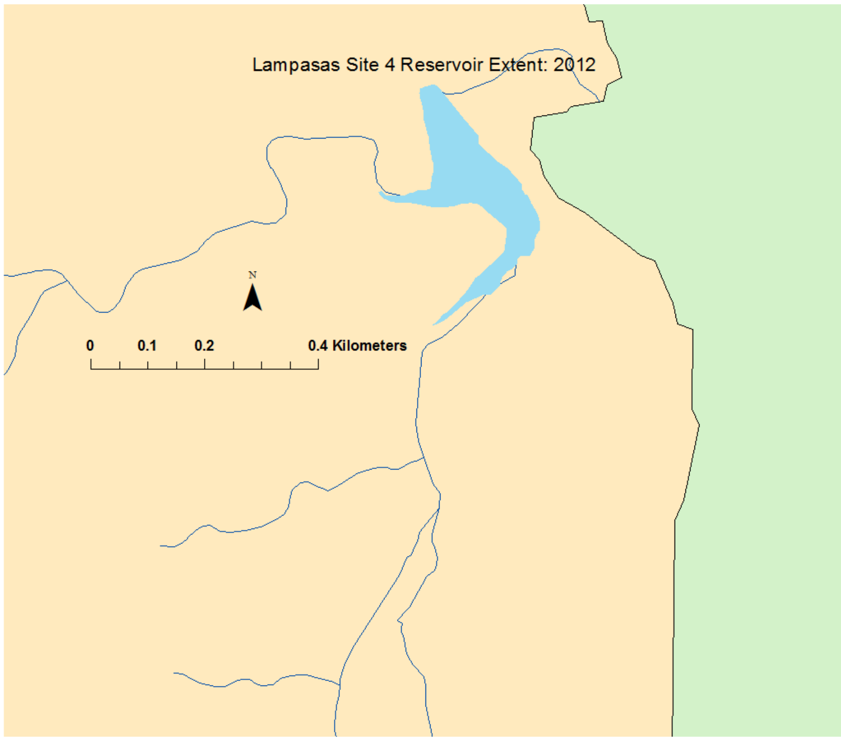

Lampasas Site 4 Reservoir Extent: 1982
N
0
0.1
0.2
0.4 Kilometers

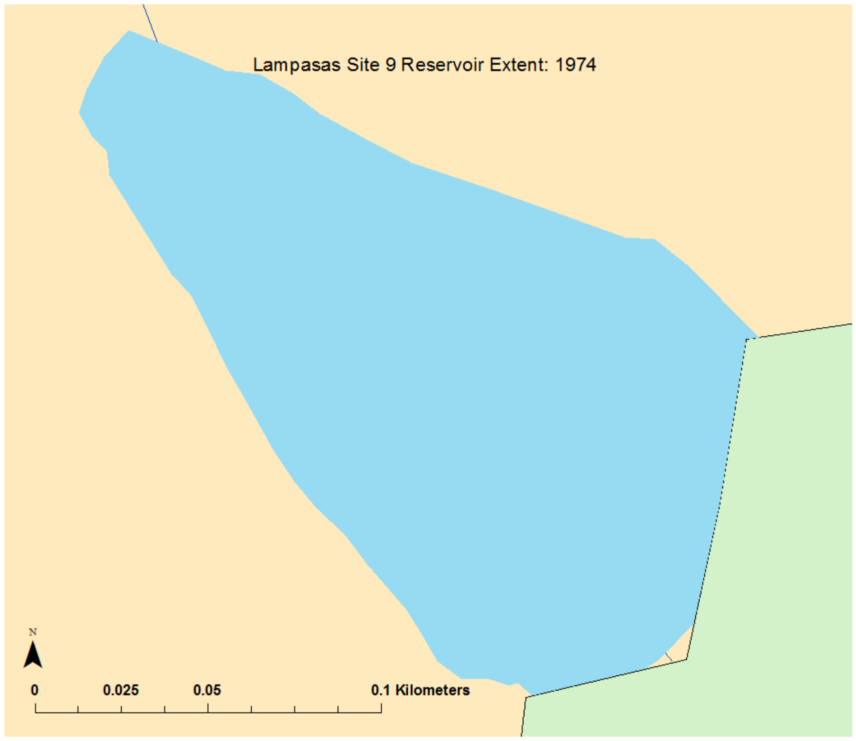

Lampasas Site 4 Reservoir Extent: 1995
N
0
0.1
0.2
0.4 Kilometers

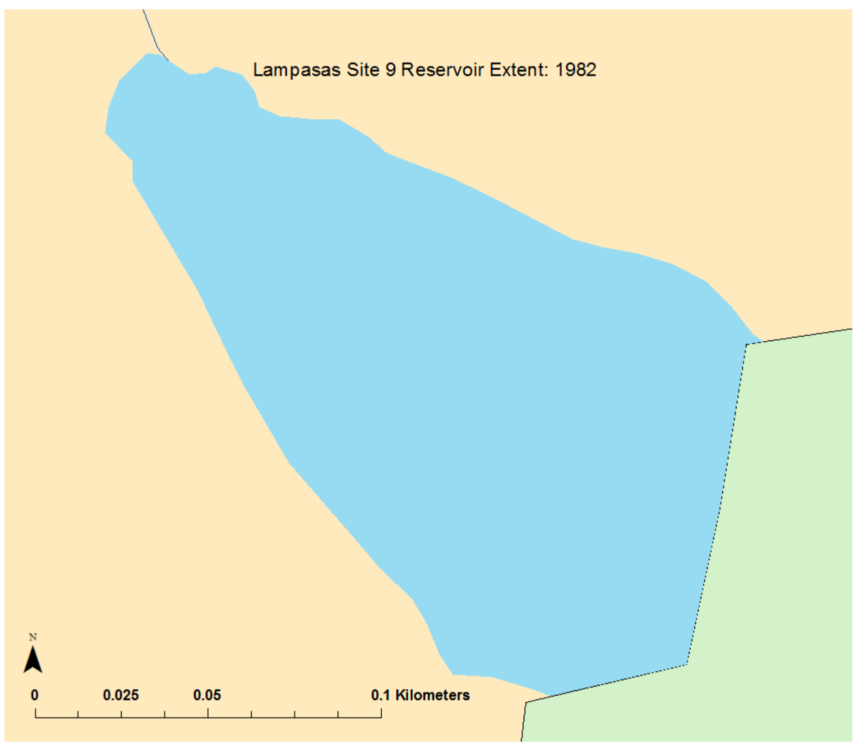

Lampasas Site 4 Reservoir Extent: 2004
N
0
0.1
0.2
0.4 Kilometers

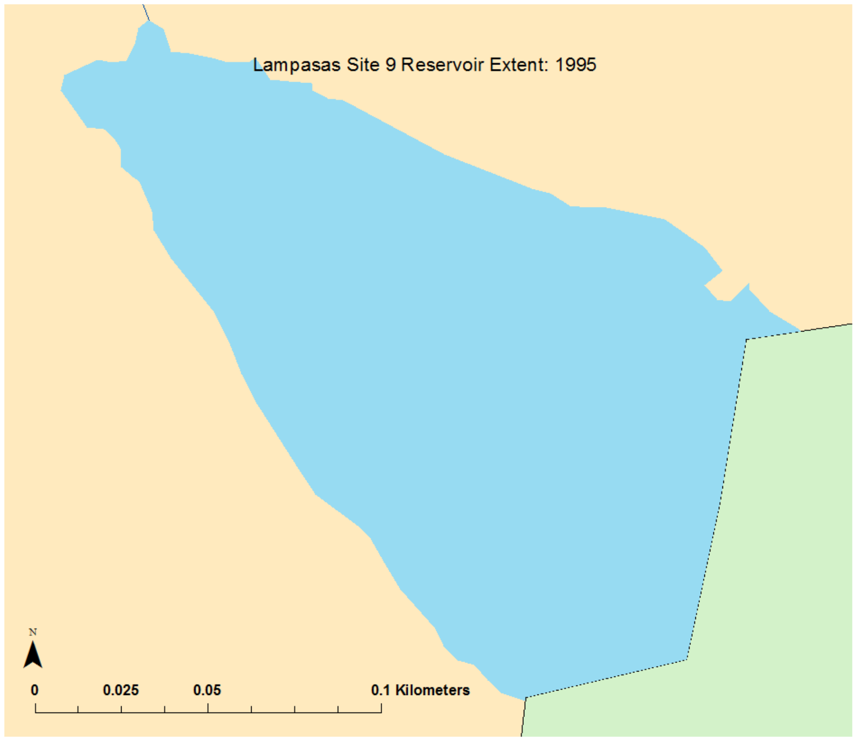

Lampasas Site 4 Reservoir Extent: 2008
N
0 0.1 0.2 0.4 Kilometers

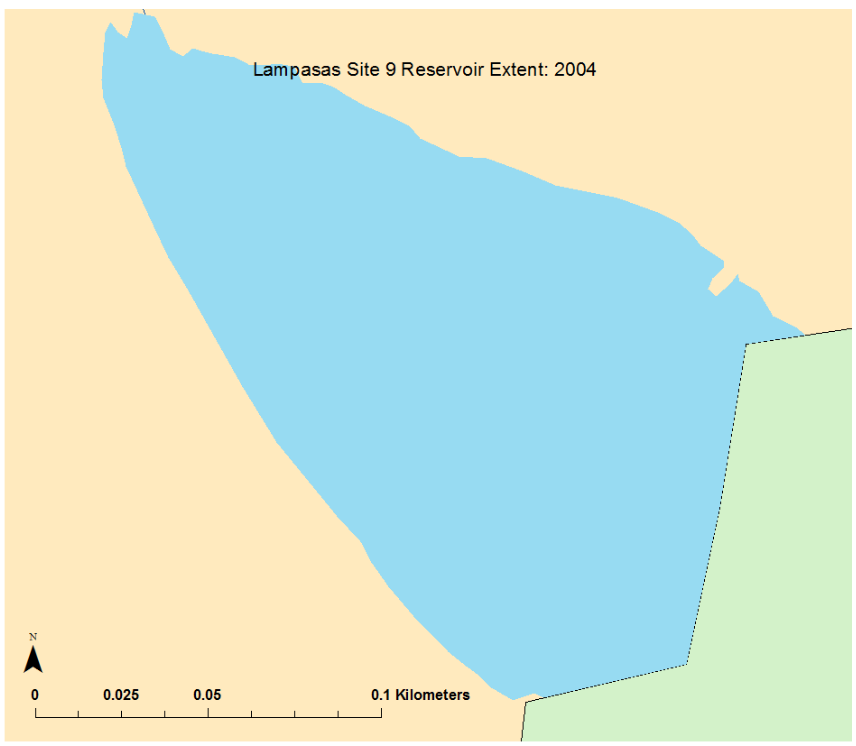

Lampasas Site 4 Reservoir Extent: 2010
N
0 0.1 0.2 0.4 Kilometers

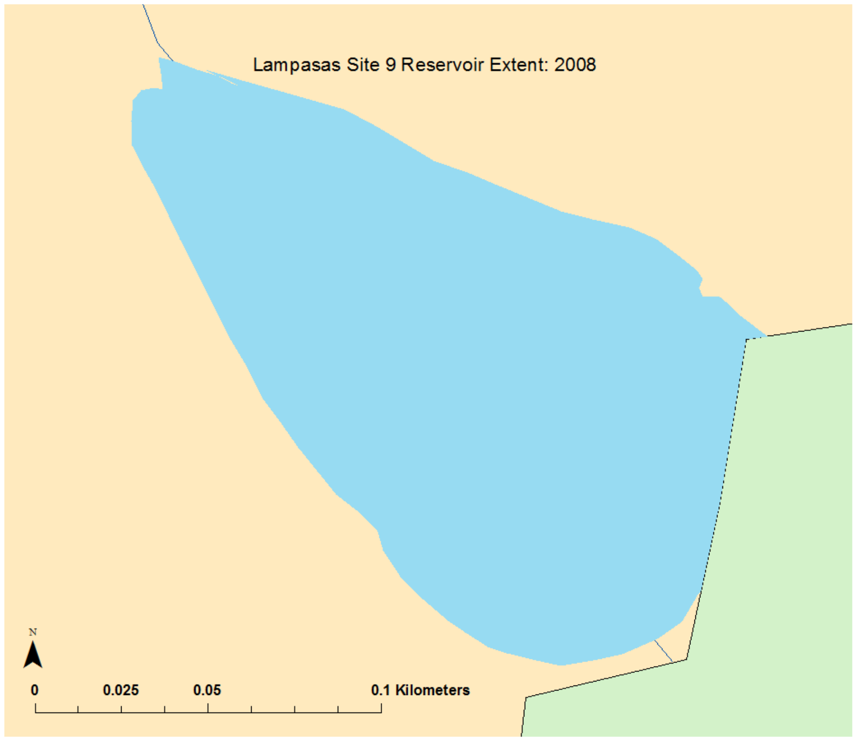

Lampasas Site 4 Reservoir Extent: 2012
N
0
0.1
0.2
0.4 Kilometers

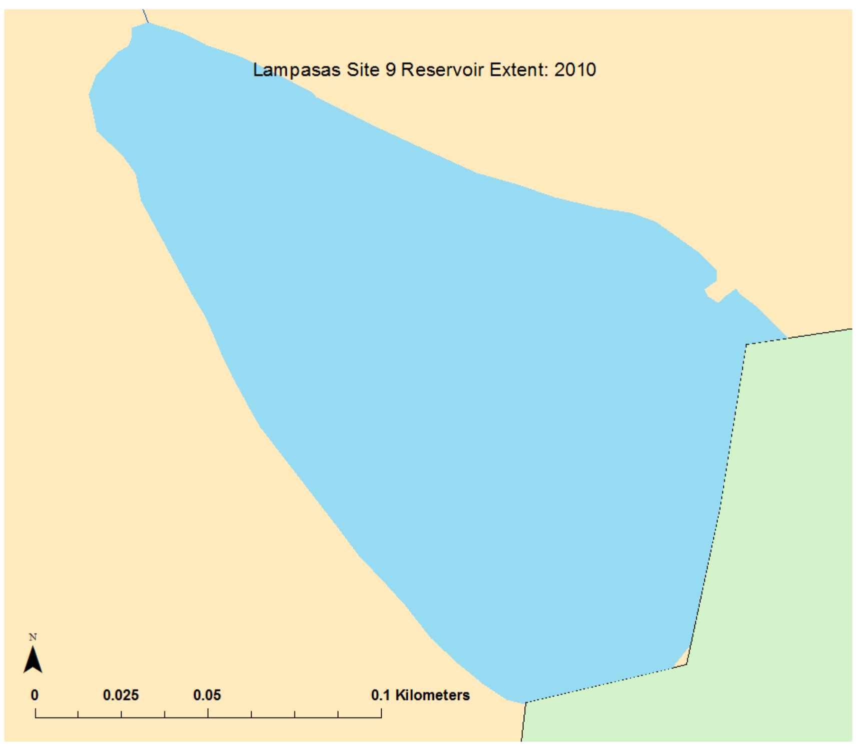

Lampasas Site 9 Reservoir Extent: 1974
N
0
0.025
0.05
0.1 Kilometers

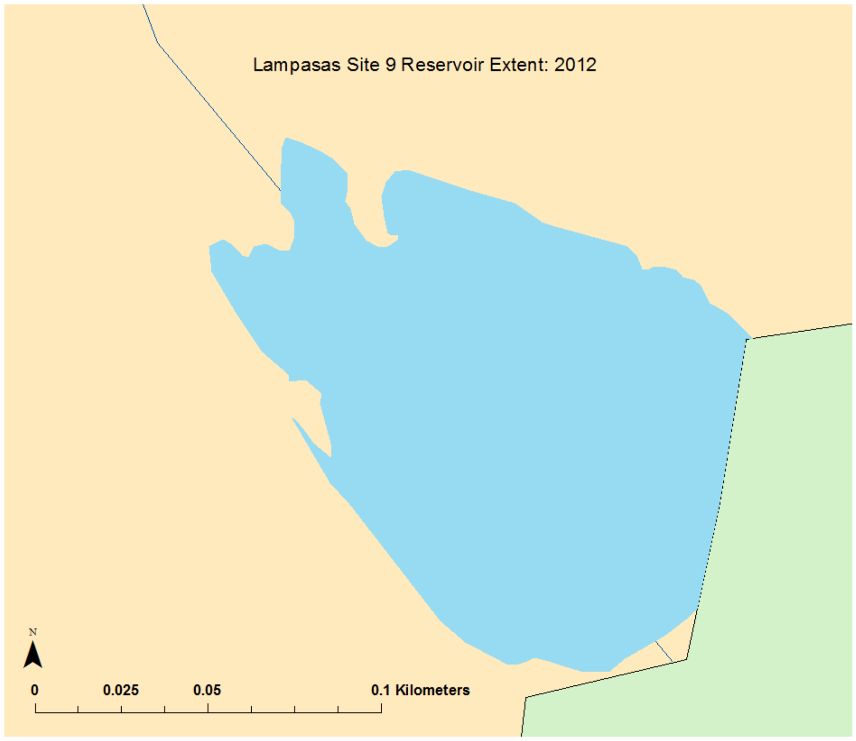

Lampasas Site 9 Reservoir Extent: 1982
N
0
0.025
0.05
0.1 Kilometers

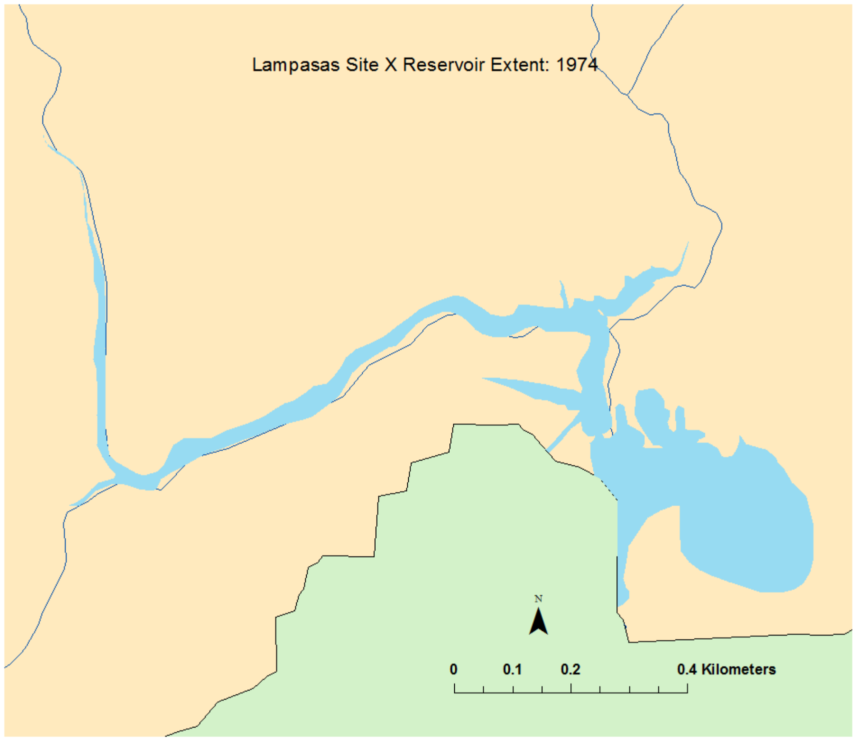

Lampasas Site 9 Reservoir Extent: 1995
N
0
0.025
0.05
0.1 Kilometers

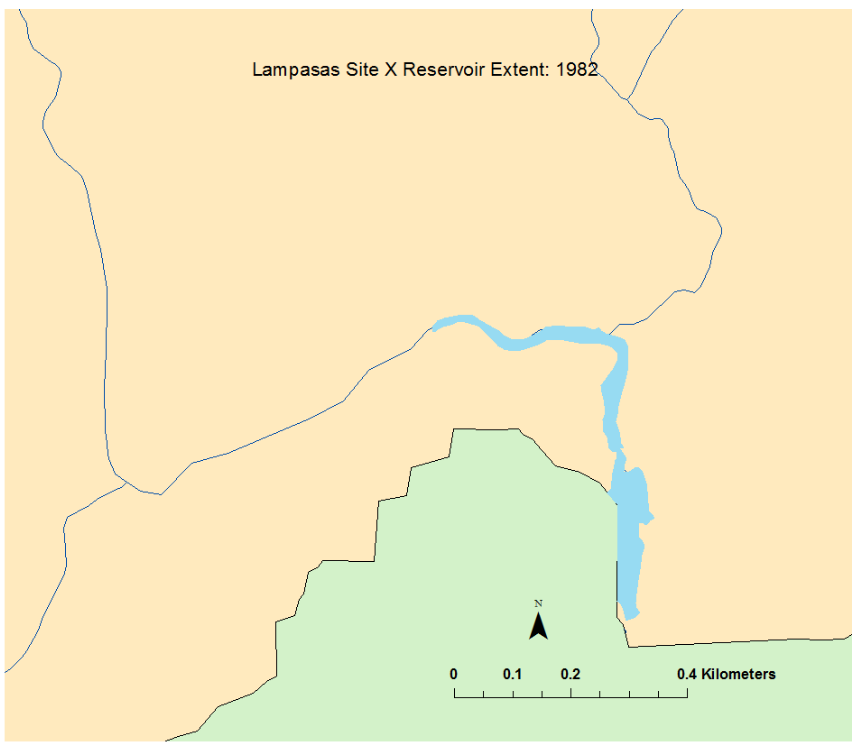

Lampasas Site 9 Reservoir Extent: 2004
N
0
0.025
0.05
0.1 Kilometers

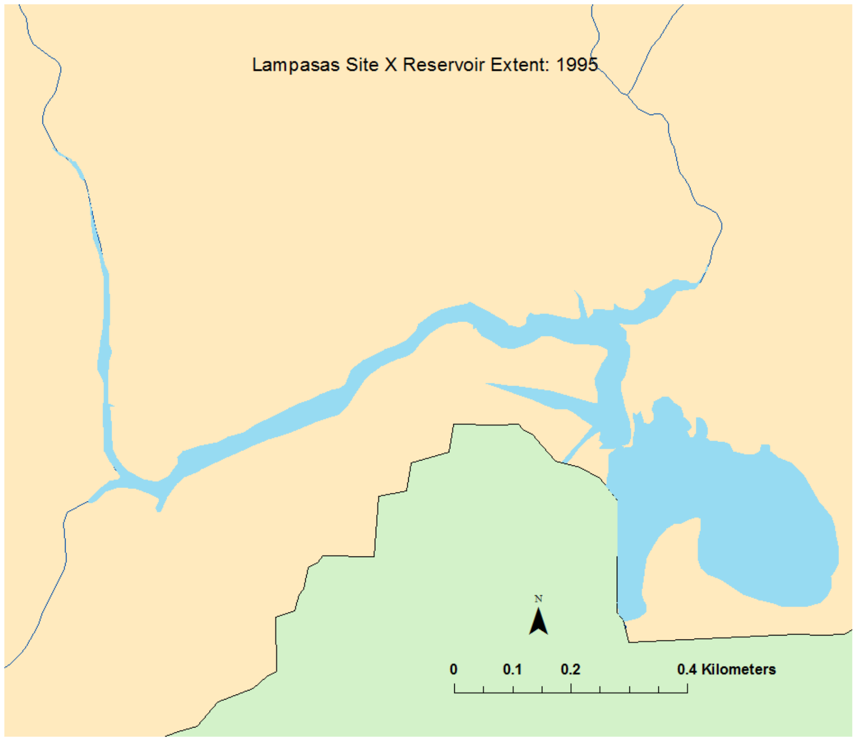

Lampasas Site 9 Reservoir Extent: 2008
N
0
0.025
0.05
0.1 Kilometers

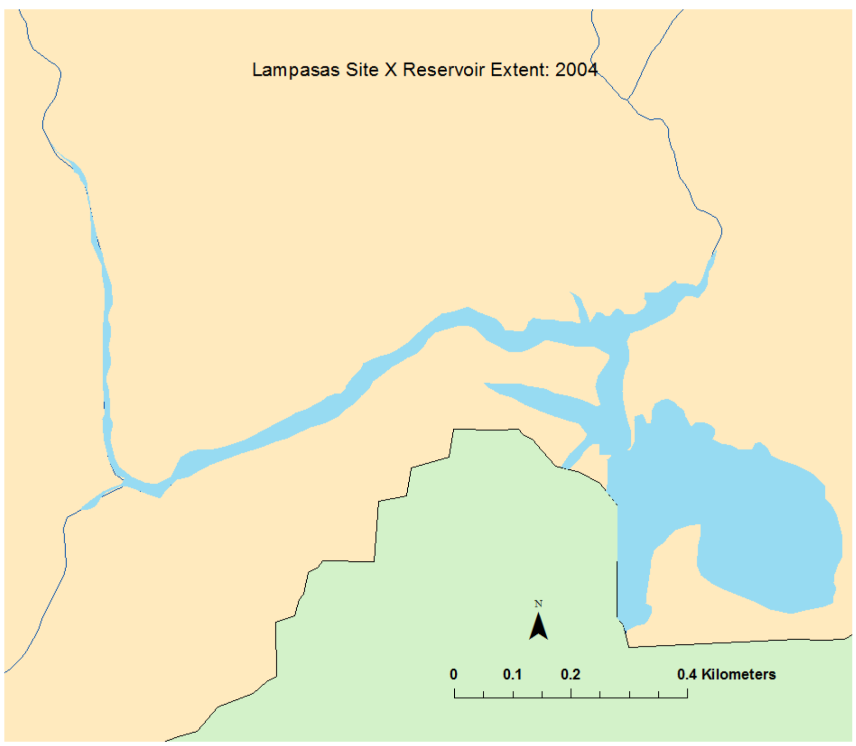

Lampasas Site 9 Reservoir Extent: 2010
N
0
0.025
0.05
0.1 Kilometers

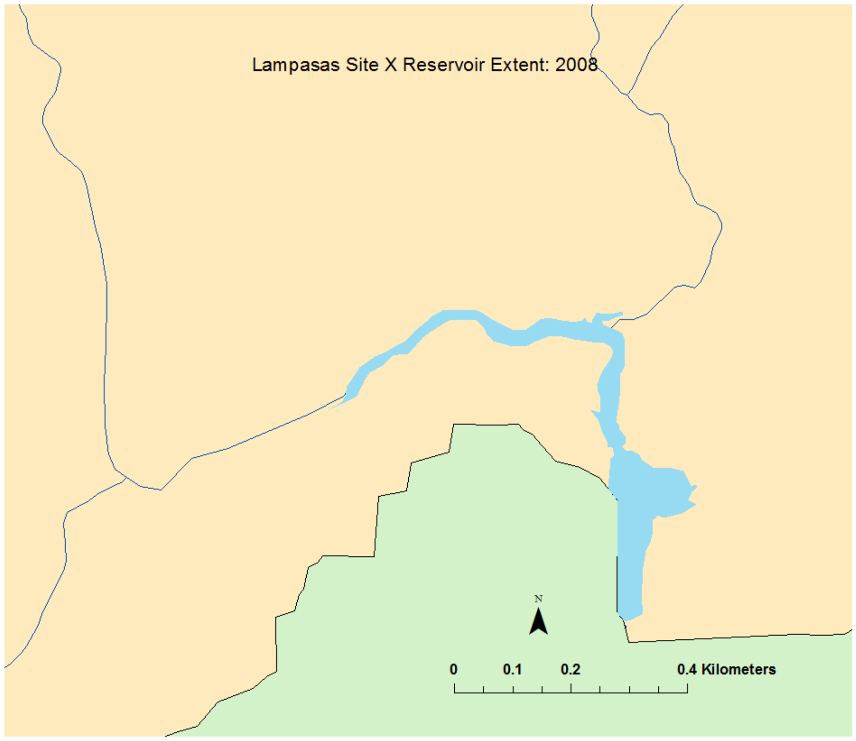

Lampasas Site 9 Reservoir Extent: 2012
N
0
0.025
0.05
0.1 Kilometers

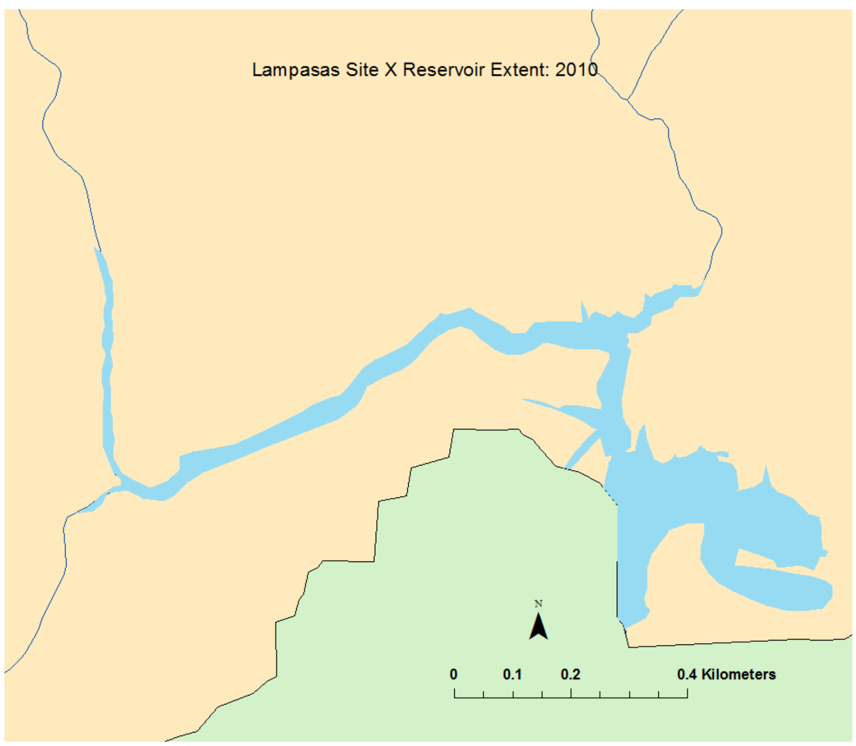

Lampasas Site X Reservoir Extent: 1974
N
0
0.1
0.2
0.4 Kilometers

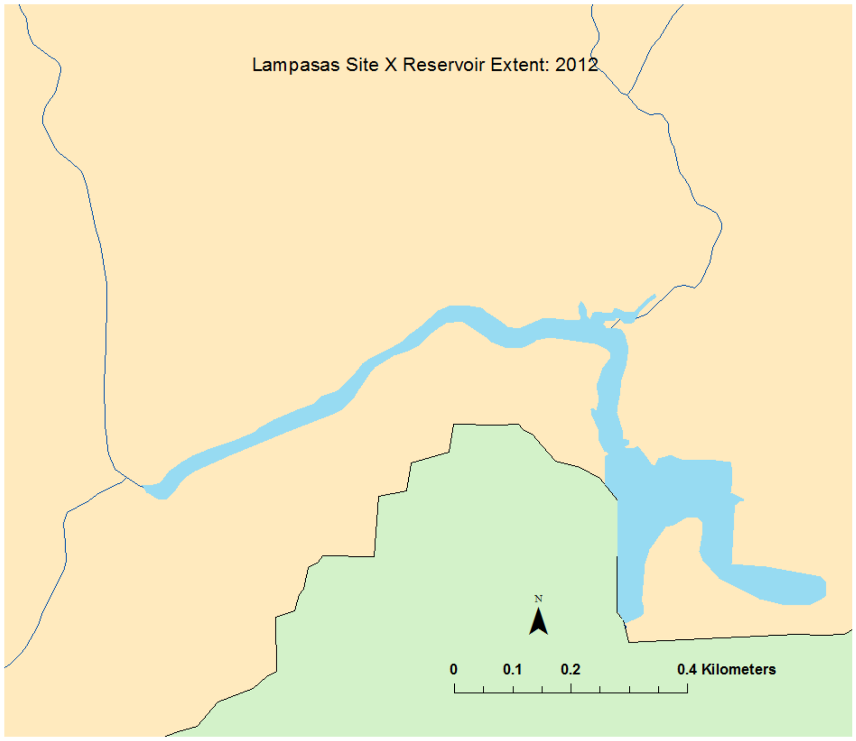

Lampasas Site X Reservoir Extent: 1982
N
0
0.1
0.2
0.4 Kilometers

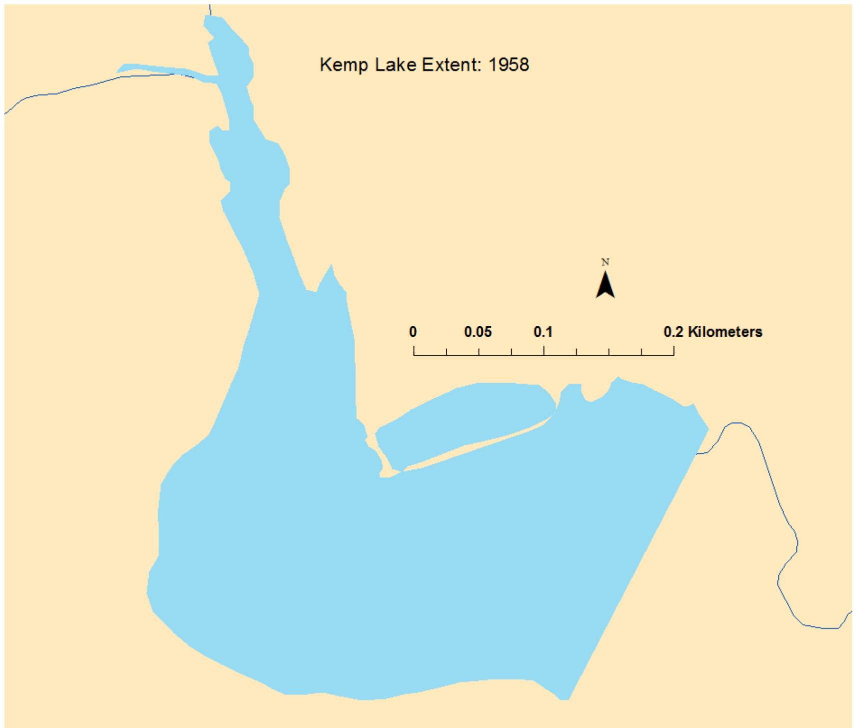

Lampasas Site X Reservoir Extent: 1995
N
0
0.1
0.2
0.4 Kilometers

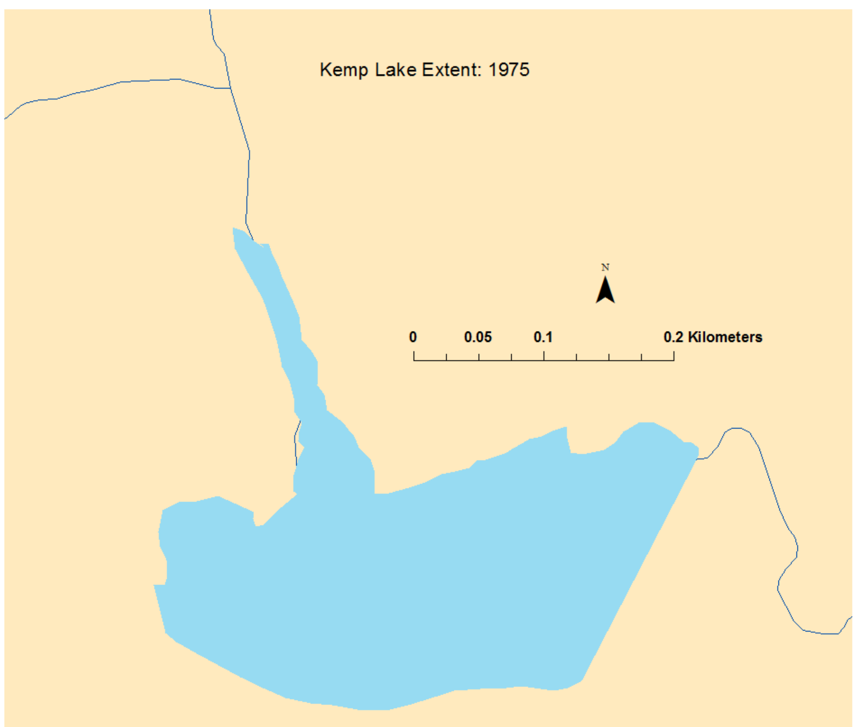

Lampasas Site X Reservoir Extent: 2004
N
0
0.1
0.2
0.4 Kilometers

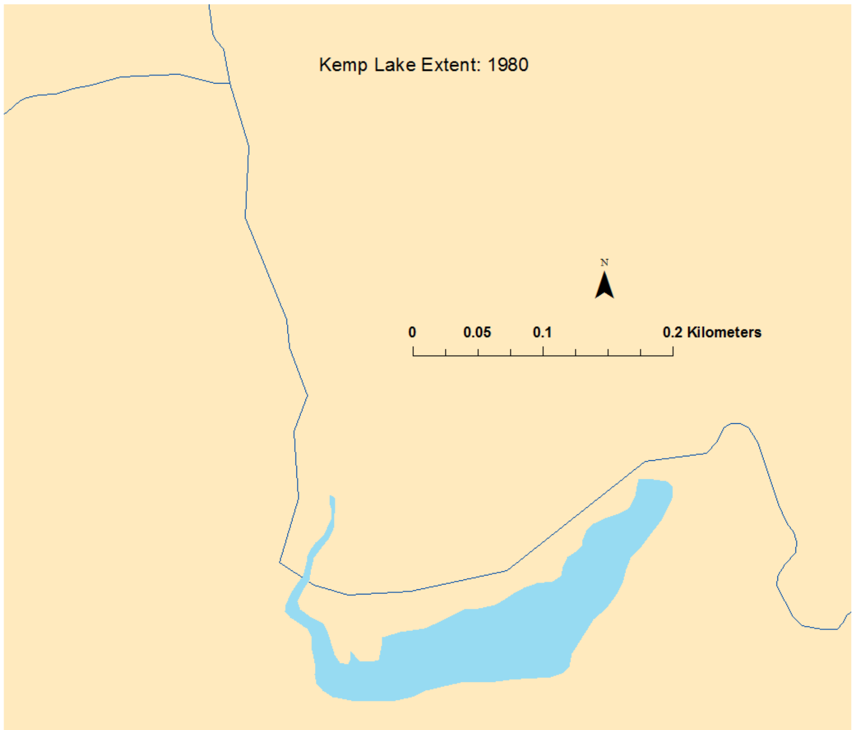

Lampasas Site X Reservoir Extent: 2008
N
0
0.1
0.2
0.4 Kilometers

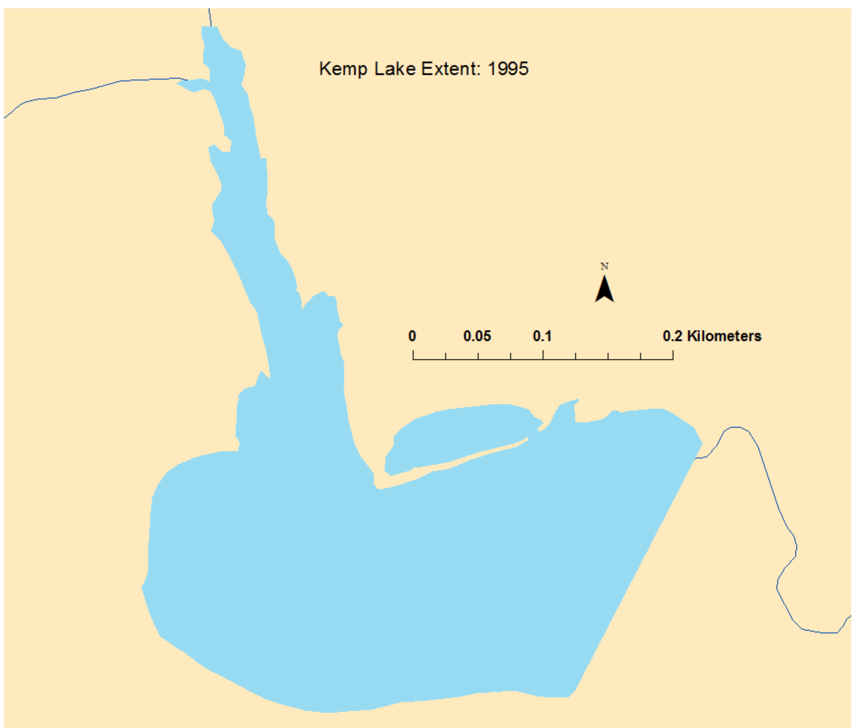

Lampasas Site X Reservoir Extent: 2010
N
0
0.1
0.2
0.4 Kilometers

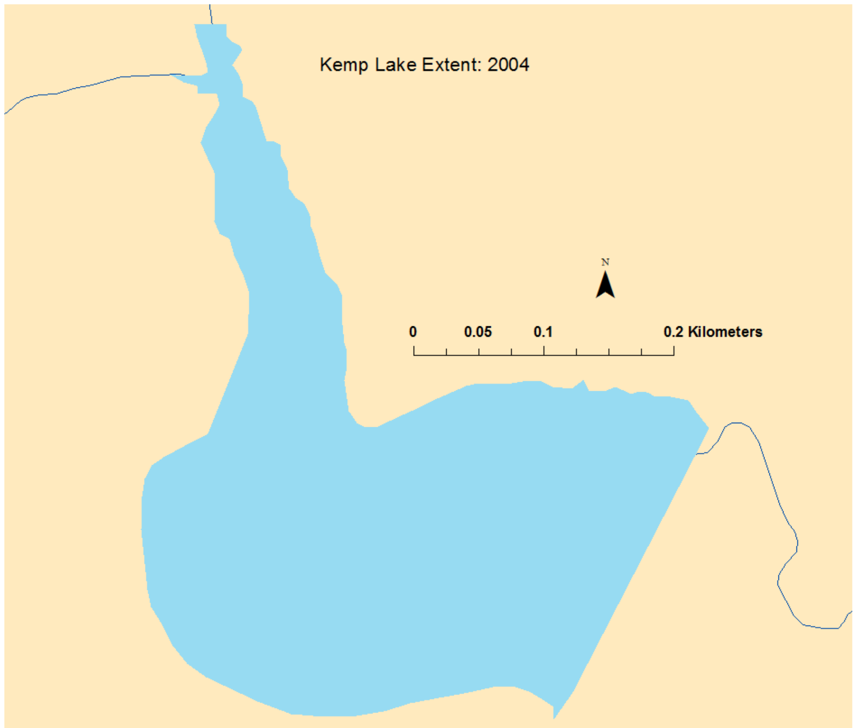

Lampasas Site X Reservoir Extent: 2012
N
0
0.1
0.2
0.4 Kilometers

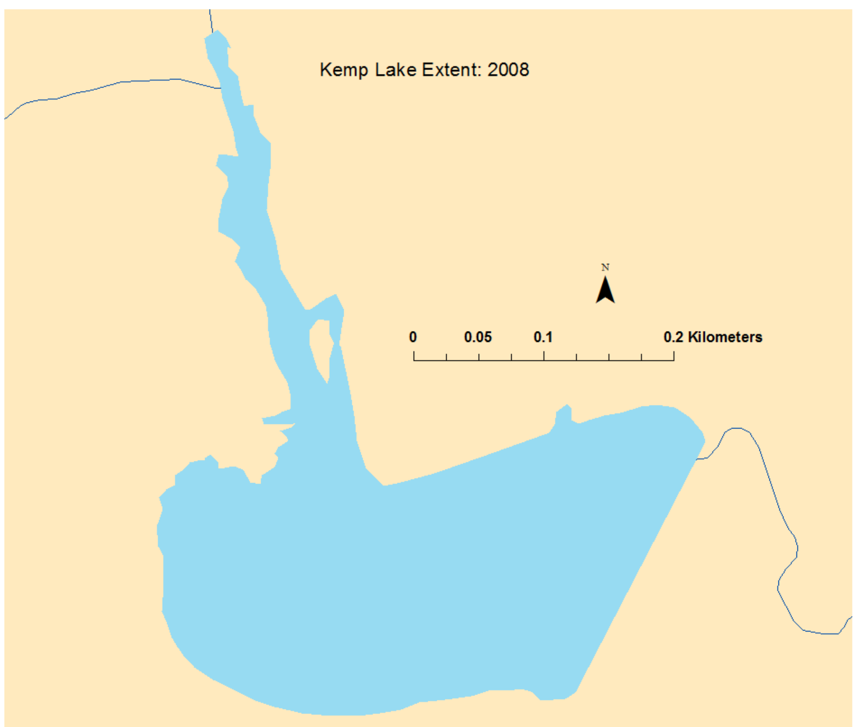

Kemp Lake Extent: 1958
N
0
0.05
0.1
0.2 Kilometers

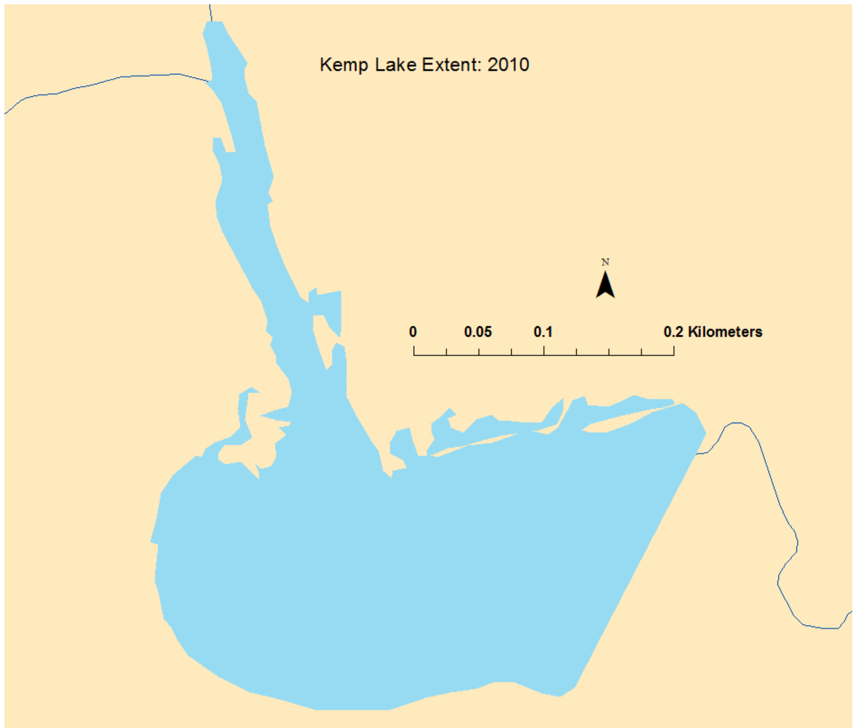

Kemp Lake Extent: 1975
N
0
0.05
0.1
0.2 Kilometers

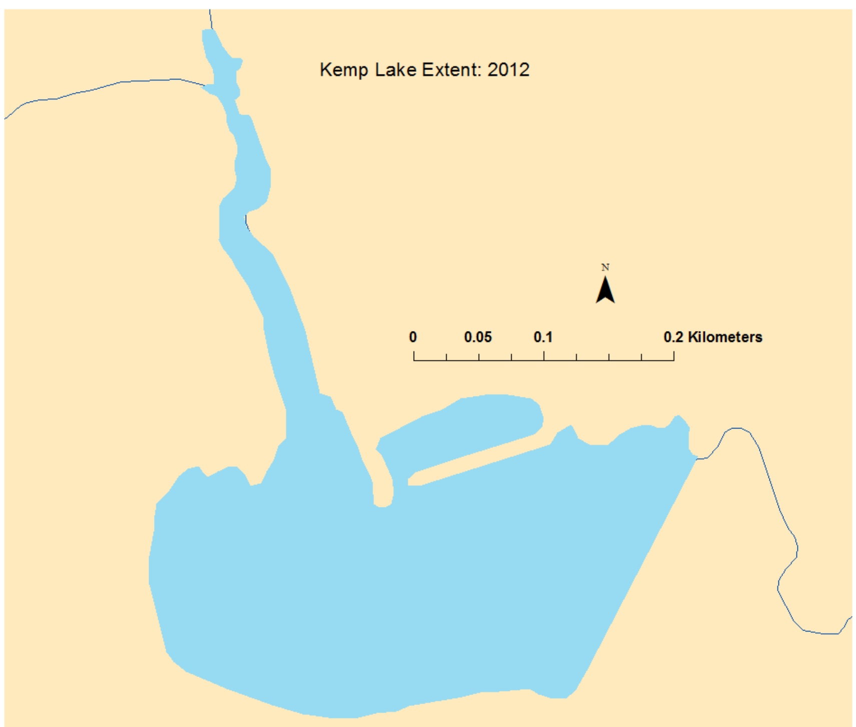

Kemp Lake Extent: 1980
N
0
0.05
0.1
0.2 Kilometers

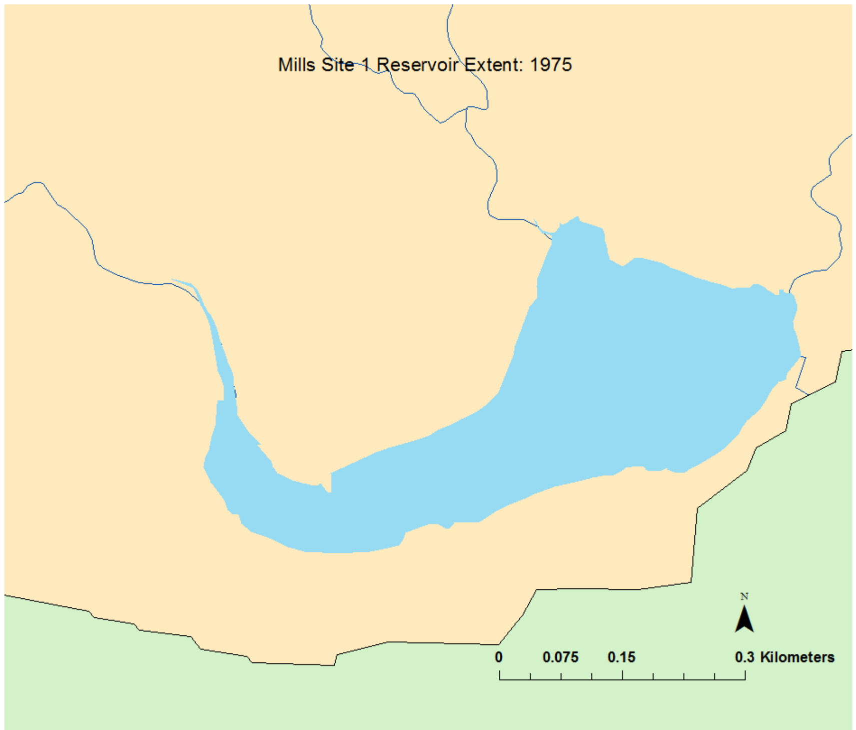

Kemp Lake Extent: 1995
N
0
0.05
0.1
0.2 Kilometers

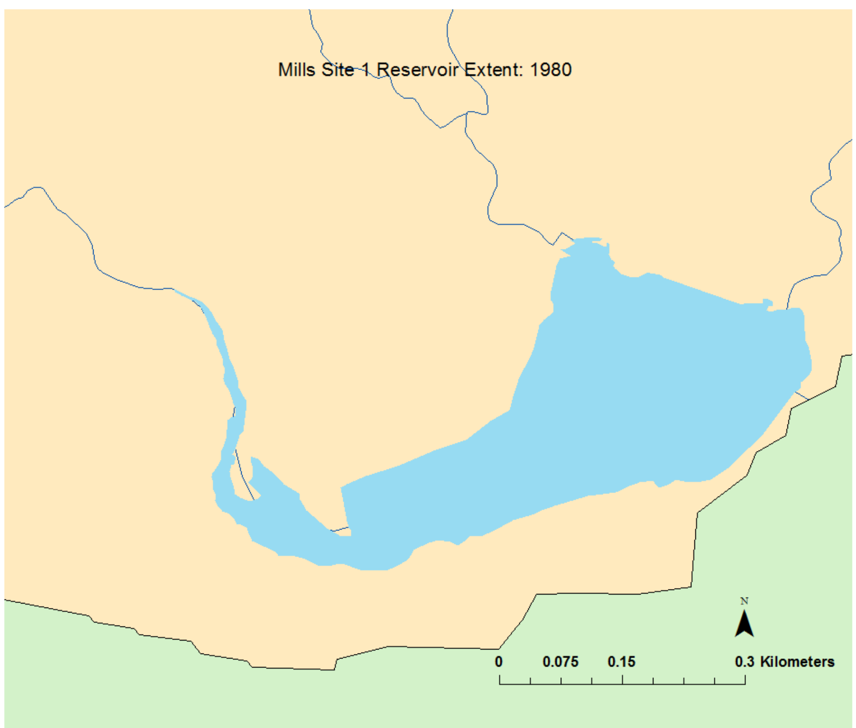

Kemp Lake Extent: 2004
N
0
0.05
0.1
0.2 Kilometers

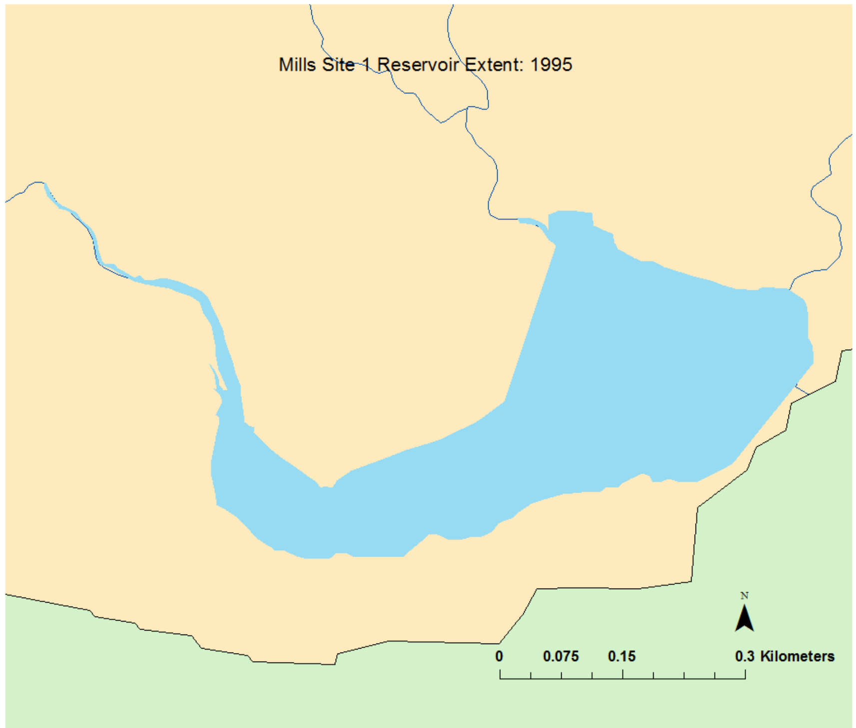

Kemp Lake Extent: 2008
N
0
0.05
0.1
0.2 Kilometers


Kemp Lake Extent: 2010
N
0 0.05 0.1 0.2 Kilometers

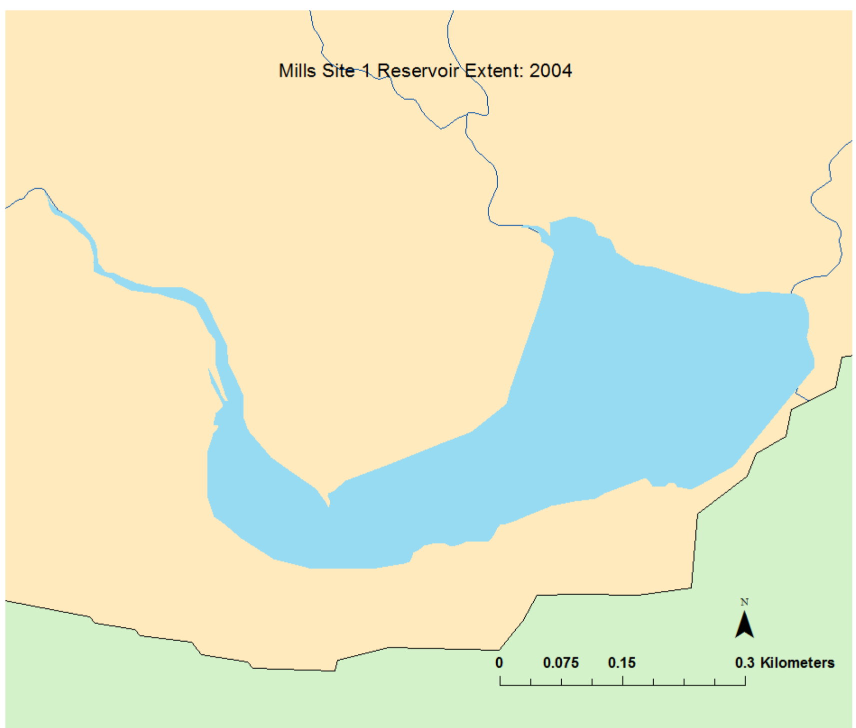

Kemp Lake Extent: 2012
N
0 0.05 0.1 0.2 Kilometers


Mills Site 1 Reservoir Extent: 1975
N
0
0.075
0.15
0.3 Kilometers

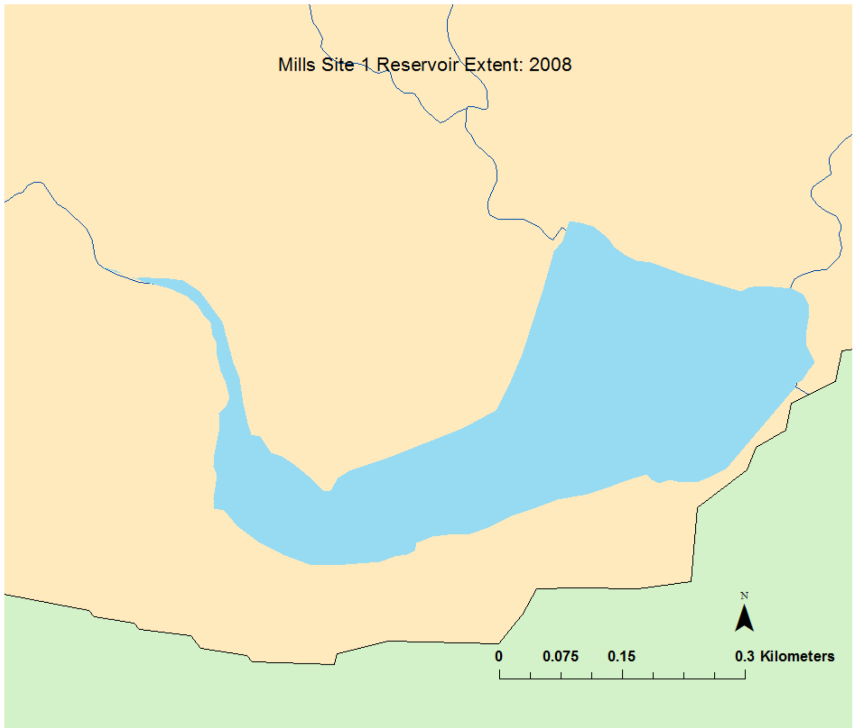

Mills Site 1 Reservoir Extent: 1980
N
0
0.075
0.15
0.3 Kilometers

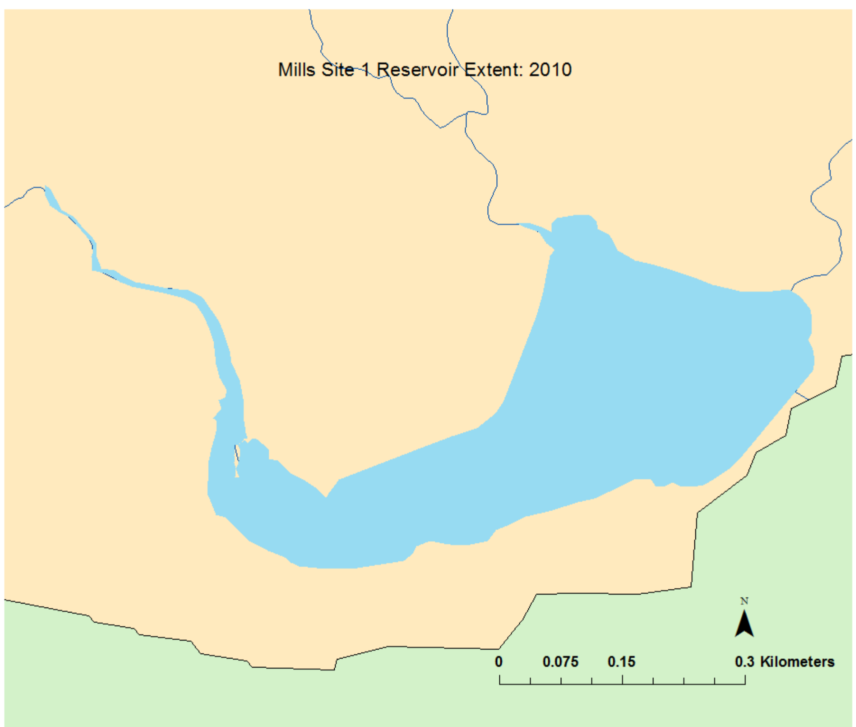


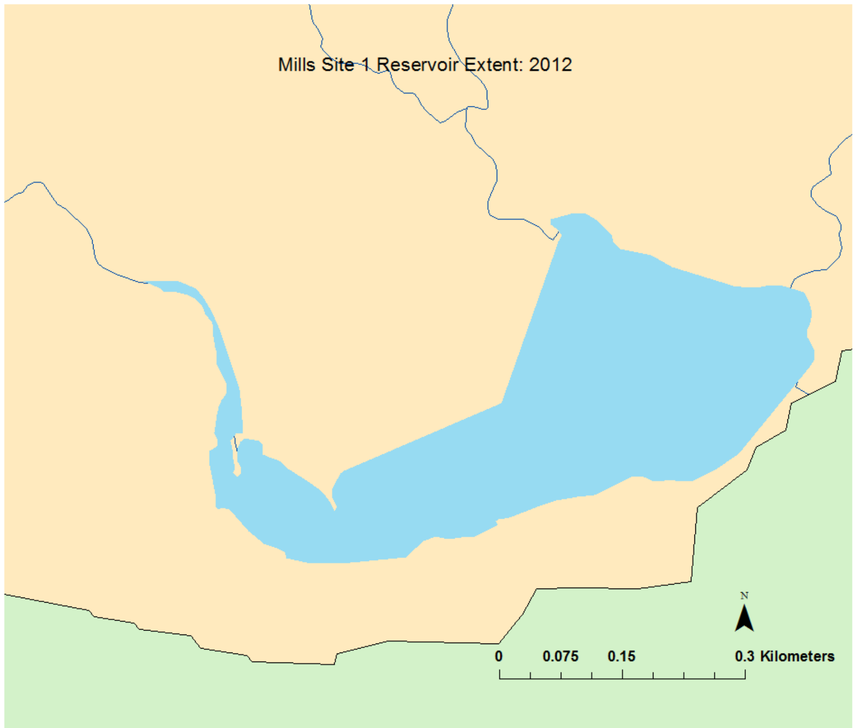
Mills Site 1 Reservoir Extent: 1995
N
0
0.075
0.15
0.3 Kilometers

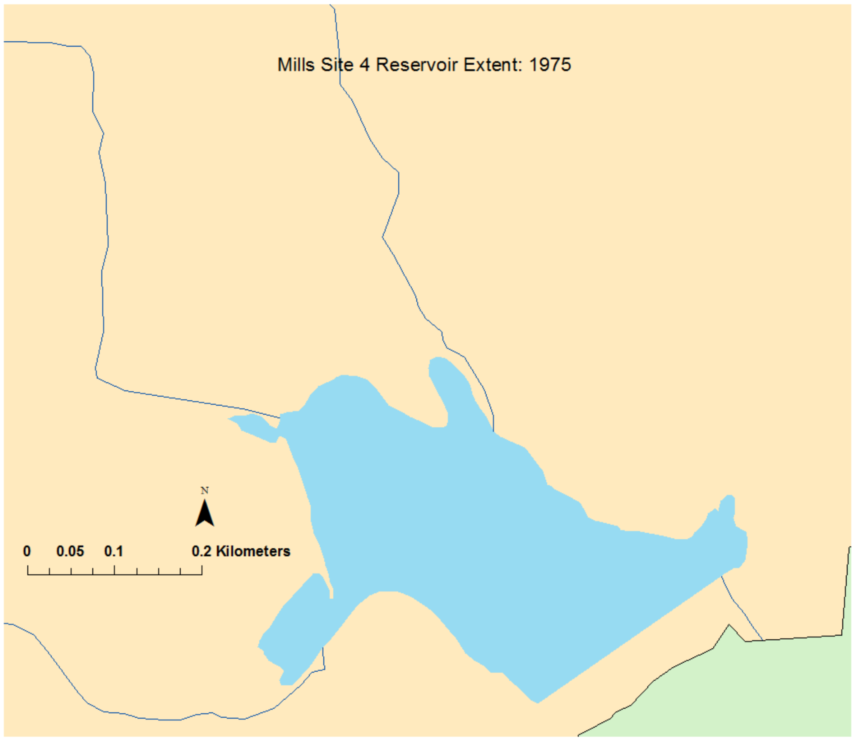


Mills Site 1 Reservoir Extent: 2004
N
0
0.075
0.15
0.3 Kilometers

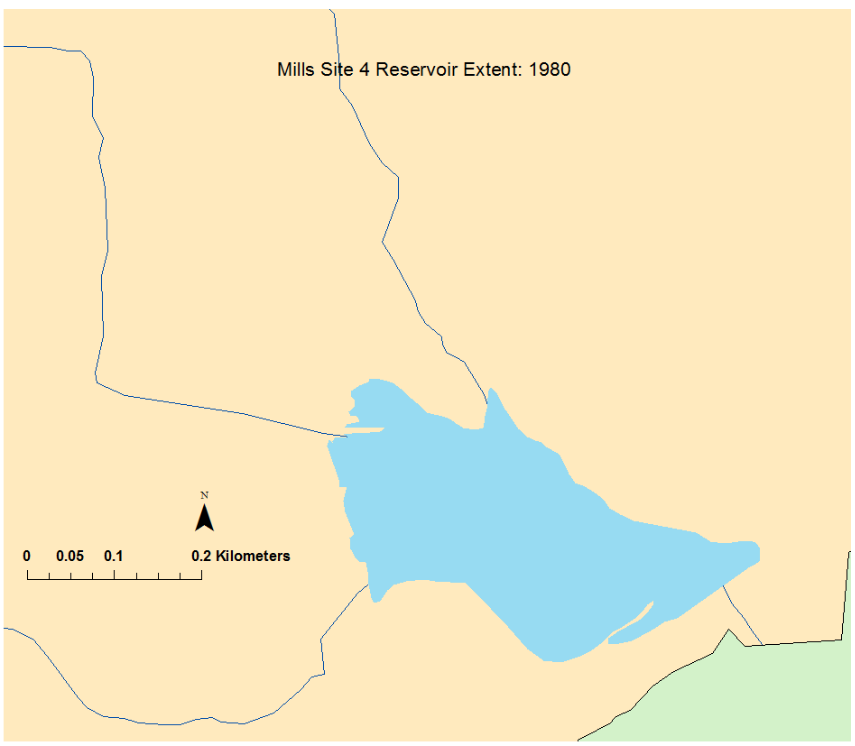

Mills Site 1 Reservoir Extent: 2008
N
0
0.075
0.15
0.3 Kilometers


Mills Site 1 Reservoir Extent: 2010
N
0
0.075
0.15
0.3 Kilometers

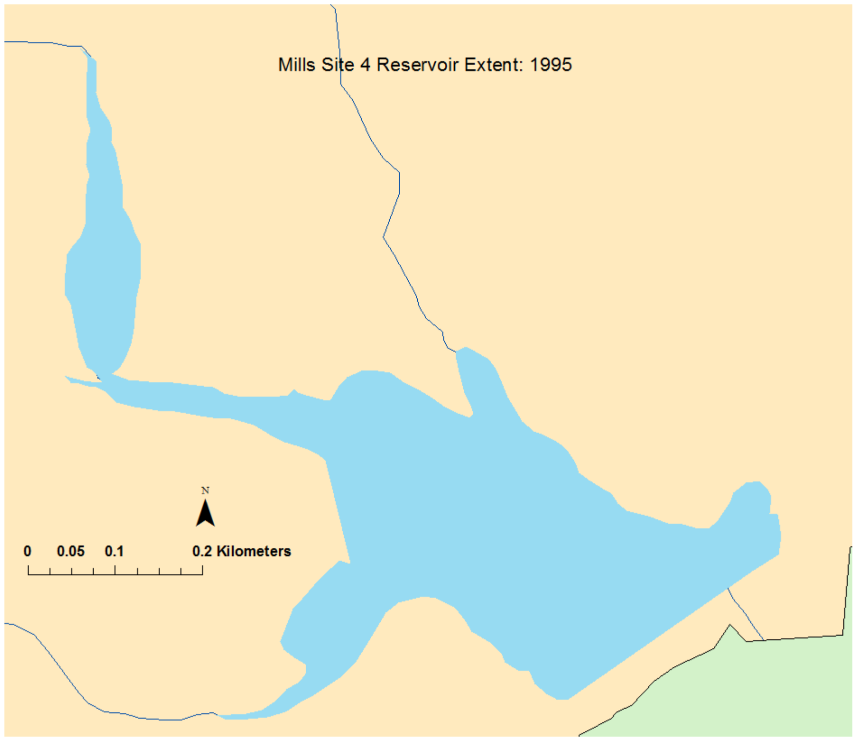

Mills Site 1 Reservoir Extent: 2012
N
0
0.075
0.15
0.3 Kilometers

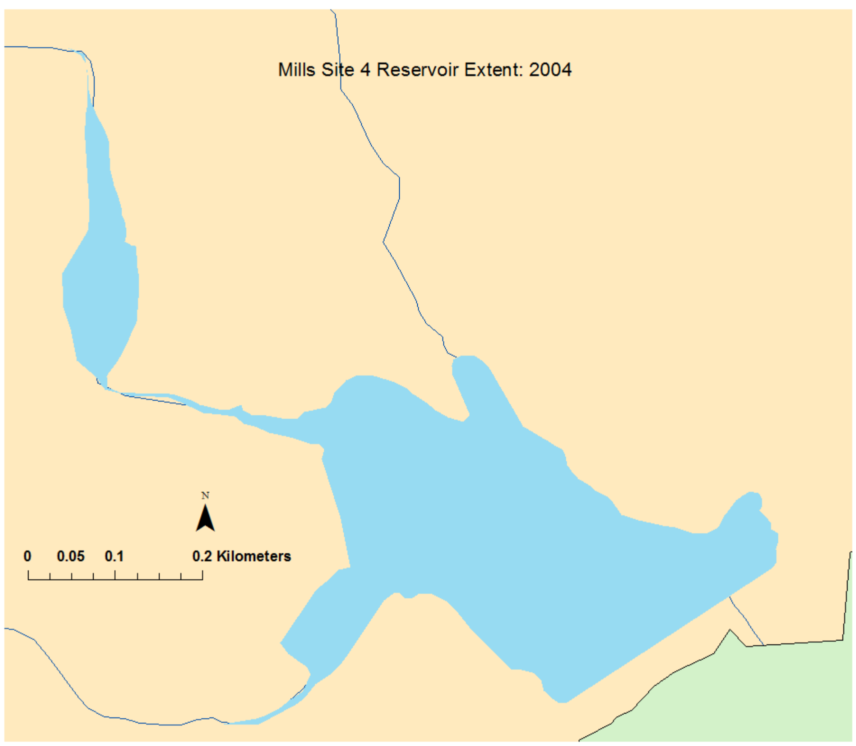


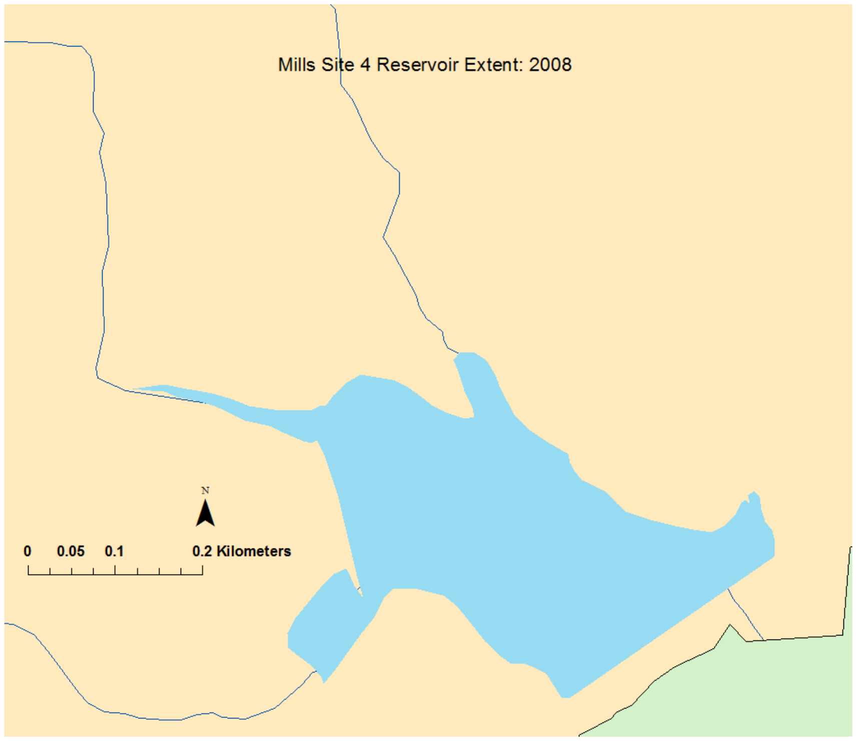
Mills Site 4 Reservoir Extent: 1975
N
0 0.05 0.1 0.2 Kilometers

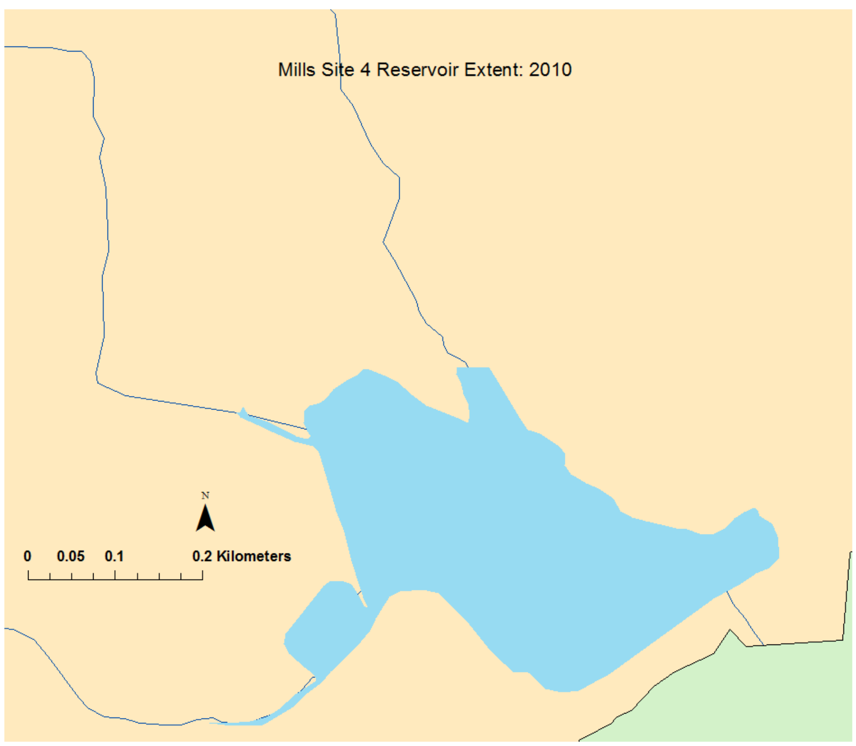


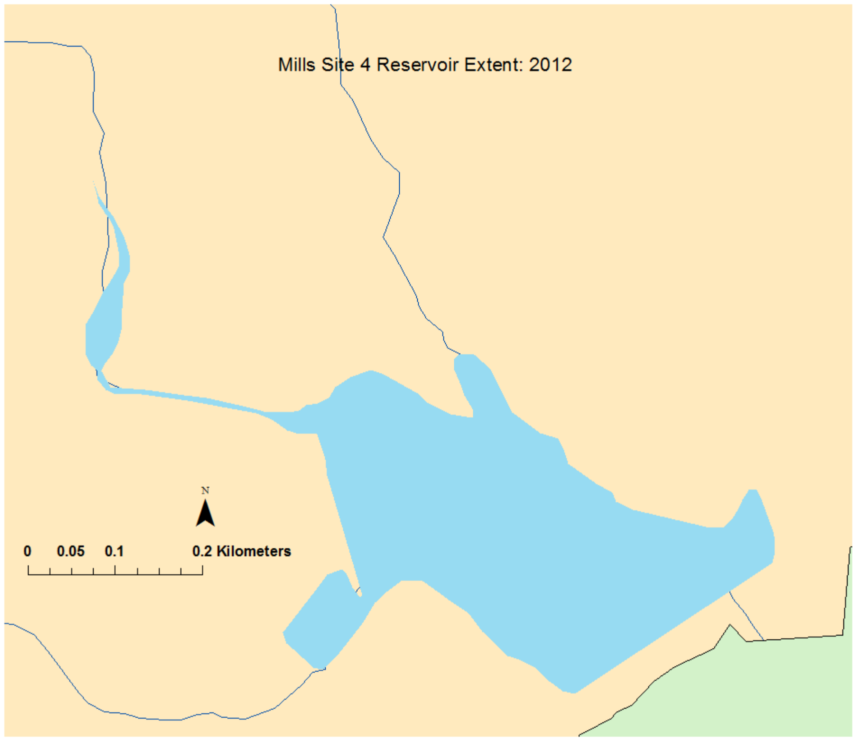
Mills Site 4 Reservoir Extent: 1980
N
0 0.05 0.1 0.2 Kilometers


Mills Site 4 Reservoir Extent: 1995
N
0
0.05
0.1
0.2 Kilometers


Mills Site 4 Reservoir Extent: 2004
N
0
0.05
0.1
0.2 Kilometers


Mills Site 4 Reservoir Extent: 2008
N
0
0.05
0.1
0.2 Kilometers


Mills Site 4 Reservoir Extent: 2010
N
0
0.05
0.1
0.2 Kilometers



Mills Site 4 Reservoir Extent: 2012
N
0
0.05
0.1
0.2 Kilometers